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# Past and present vegetative and wildlife diversity in relation to an existing reserve network : a GIS evaluation of the Seeley-Swan landscape northwestern Montana

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PAST AND PRESENT VEGETATIVE AND WILDLIFE DIVERSITY IN RELATION TO AN EXISTING RESERVE NETWORK:

A GIS Evaluation of the Seeley-Swan Landscape, Northwestern Montana

by

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A landscape-level analysis of biodiversity was conducted using a geographic information system (GIS). Existing vegetation was classified and labeled according to cover type from Landsat TM imagery acquired 20 July 1991; comparisons were then drawn with a historic vegetation layer digitized from 1930s maps. For 20 wildlife species, predicted habitat was mapped for the 1930s and 1990s based on vegetation, topography, proximity to water, road density, and other variables available in the GIS database. Since the 1930s, the Seeley-Swan landscape has become increasingly fragmented, and proportions of individual cover types have shifted as timber harvest has replaced fire as the dominant disturbance process. In particular, mature/overmature forests, the landscape's matrix component in the 1930s, have declined in total area, while seedling and sapling seral stages have become more extensive and could potentially replace mature/overmature forests as the landscape matrix. This shift is reflected in habitat predictions for wildlife species using older forests; in general, habitat has declined in total area and become more fragmented in its configuration.

Cover types, elevation zones, and biophysical zones currently underrepresented in the existing network of protected areas were identified, and all sites were scored for potential inclusion in the reserve network. Although a substantial proportion (29%) of the landscape is already accorded high protection, the lower elevations (< 1600 m) and associated cover types and biophysical zones are poorly represented in the existing reserve network. Inclusion of low-elevation, old-growth forests -- particularly ponderosa pine, western red cedar, and extensive stands of mixed conifer composition, such as those blanketing the valley floor in the 1930s -- would improve the existing network. In addition, small reserves proposed to protect sensitive plants could be expanded to minimize outside influences and increase the probability of these reserves playing a functional role in maintaining healthy ecosystem and landscape processes. The process of augmenting the existing network of protected areas is complicated by the number of landowners involved. Key players include the Flathead National Forest and Plum Creek Timber Company; Lolo National Forest, the Montana Department of State Lands, and many individual landowners also will play important roles.

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## Chapter 1:
## Introduction.

*"To keep every cog and wheel is the first precaution of intelligent tinkering." (Leopold 1953:147)*

Disturbance is a normal characteristic of natural landscapes: Throughout time, landscape patterns have been shaped by disturbance processes. However, human tinkering (intelligent or otherwise) has greatly accelerated levels of disturbance in some landscapes and suppressed them in others. The resultant destruction and degradation of habitat has placed many species in jeopardy. Thus, conservation of biodiversity in these modified landscapes has become a major concern (Wilson 1988).

Biodiversity is "the variety and variability among living organisms and the ecological complexes in which they occur" (OTA 1987), the diversity of life in all its forms and at all levels of organization, from molecules to ecosystems (Hunter 1990). The "crisis discipline" of conservation biology (Soule 1985) grew out of concerns over rapid loss of biodiversity, and "...addresses the biology of species, communities, and ecosystems that are perturbed, either directly or indirectly, by human activities or other agents" (Soule 1985:727). In tackling topics of such broad scope, we are quickly led to ask questions of equally broad scale about anthropogenically caused changes in the structure and function of ecosystems and landscapes. Landscape ecology, another young and expanding discipline, allows us to formalize these broad-scale questions by studying landscape structure, function, and change, then apply these principles to real-world problems (Forman and Godron 1986). In turn, geographic information systems (GIS) provide the technology to address broad-scale questions, however crude the preliminary results may be. A GIS is a powerful set of tools for collecting, storing, retrieving, transforming, and displaying spatial data (Burrough 1986). As the capabilities of the GIS toolbox develop, a related GIS discipline is rapidly evolving.

Techniques for the evaluation of biodiversity, like gap analysis (Scott et al. 1993) and representativeness assessment (Austin and Margules 1986), merge the disciplines of conservation biology, landscape ecology, and GIS analysis. Gap analysis is a coarse-filter approach, meant to provide a relatively quick overview of the distribution and conservation status of various components of biodiversity by focusing on native terrestrial vertebrates. In this process, GIS map layers are overlaid to identify the individual species, species-rich areas, and vegetation types that are not adequately represented in areas currently managed for biodiversity (Scott et al. 1993). Similarly, representativeness assessment is a means of evaluating how well a reserve or system of reserves represents the range of biological variation in a region (Austin and Margules 1986). It emphasizes biophysical characteristics like climate and landform types rather than the vertebrate species targeted in gap analysis.

While promising, these interdisciplinary techniques are still in the developmental phase, and are in need of extensive exploration and validation. No existing technique on its own can address all the questions that must be answered in an effective conservation strategy. Thus, I attempt to combine various techniques for a small landscape, the Seeley-Swan, to present a comprehensive view of biological diversity and its protection in the area. My objectives are to:

1) Compare historic and current landscape characteristics in order to evaluate deviation from natural vegetative conditions.

2) Assess landscape diversity by characterizing both common landscape elements and unique areas ("hot spots" of richness, or centers of endemism), with emphasis on old growth and sensitive species.

    a) Identify areas of high species richness and their protection status through gap analysis: overlay maps of vertebrate and sensitive plant species distributions with maps of land ownership and use.

b) Conduct a representativeness assessment, focusing on vegetative communities, to complete the landscape characterization.

3) Propose a natural areas network based on a variety of factors, including connectivity, representativeness, and species richness; again, give special attention to old growth, unique habitats, and sensitive species.

4) Attempt to identify landscape indicators of diversity in order to aid in future biodiversity assessments.

5) Outline a process or formula for evaluating biodiversity at the landscape level.

My premise is simple and obvious: the Seeley-Swan landscape has changed over the last 50 years due to human activities. While obvious, this change is worth documenting and quantifying. We have to know what *was* there to assess what has been lost, and we have to know what *is* there to decide what to protect.

My emphasis is on exploring a process for evaluating biodiversity as well as obtaining results for this particular study area. The Seeley-Swan, with its natural mosaic of forest and wetland complexes and its complex ownership patterns, is an ideal area for landscape-level assessments. Although limited in scope, my study will: 1) provide information on threatened, endangered, and sensitive species, and 2) serve as an application of both reserve design theory and ecosystem management principles. Thus, despite the emphasis on process, the study is still of considerable practical value.

Chapter 2 describes the Seeley-Swan landscape, including the GIS map layers that represent the study area in the digital world. Chapter 3 then explains how the digital vegetation layers were constructed, and how they have been compared to draw inferences about landscape change over the past 50 years. The influence of mapping resolution on landscape characterization is also explored. In chapter 4, the process of wildlife habitat modeling is addressed, with examples presented for twenty vertebrate species of special concern in Montana. Next, chapter 5 outlines the methods used to assess how well the currently designated reserves represent the vegetation in the study area. Results are synthesized into recommendations toward an ideal reserve network in the Seeley-Swan landscape.

## Chapter 2:
## Description of the Seeley-Swan.

The Seeley-Swan study area occupies approximately 247,900 ha in northwestern Montana, about 50 km northeast of Missoula. The area is about 80 km long and 20-45 km wide (Fig. 2-1), and is bordered by the Mission Mountains on the west and the Swan Range on the east. The Seeley-Swan is composed of two valleys separated in their upper reaches by a gentle divide. From the Clearwater Divide, the Swan River flows north through Swan Lake and finally into Flathead Lake; the Clearwater River flows south through a chain of lakes and eventually joins the Blackfoot River at Clearwater Junction. Elevation of the valley floor ranges from 925 to 1230 m, and the adjacent mountain peaks reach 2150-3130 m.

The area has been sculpted by a combination of continental and alpine glaciation. Lobes of the Cordilleran ice sheet moved southward through the Rocky Mountain Trench between 10,000 and 140,000 years ago (Hansen et al. 1991). One lobe advanced through the Swan Valley, spilling over the Clearwater Divide and reaching past Clearwater Junction to the south. Local mountain glaciers also advanced during the Pleistocene (Antos and Habeck 1981; Johns 1970). Beneath a thick layer of till deposited on the valley floors by these glaciers are sedimentary rocks of the Belt Series (Johns 1970). Bedrock in the Swan Range is predominantly argillite, whereas the Mission Mountains are comprised of limestone (Antos and Habeck 1981). Soils throughout the study area generally show poor profile development. Another legacy of glaciation is the Seeley-Swan's complex micro-topography of wetlands intermingled with upland terrain; distribution patterns of forest communities have been heavily influenced by this landscape complexity (Freedman and Habeck 1985).

Except for its wetland and riparian sites, the study area is blanketed by coniferous forest. Major tree species include western larch (*Larix occidentalis*), Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), western white pine (*Pinus monticola*), western red cedar (*Thuja plicata*), grand fir (*Abies grandis*), Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and whitebark pine (*Pinus albicaulis*). Representative habitat types (Pfister et al. 1977) include *Abies grandis/Clintonia uniflora* (ABGR/CLUN), *Abies lasiocarpa/Xerophyllum tenax* (ABLA/XETE), and *Pseudotsuga menziesii/Symphoricarpos albus* (PSME/SYAL).

In general, the Seeley-Swan contains highly productive sites. Some of this may be attributed to a maritime influence on its climatic regime. Precipitation averages a moderate 60-70 cm in the Swan Valley, but is much higher (200-250 cm) at high elevations. Temperatures average -15° C in winter and 28° C in summer. (USDA:FS 1985)

Although the vegetation of the Seeley-Swan was described above as a blanket of forest, a patchwork quilt might be a more apt description. Natural disturbances at various spatial and temporal scales -- glaciation in the long-term, and fire in the short-term -- created a mosaic of forest and wetlands. A variety of timber harvest prescriptions, superimposed on this naturally complex pattern, result in a highly fragmented landscape, a condition made strikingly obvious by satellite imagery.

The Seeley-Swan's checkerboard pattern of land ownership and management is another feature easily noted from Landsat TM imagery. Federal, state, corporate, and private lands are highly intermingled, especially in the valley bottom. Major landowners include the Flathead and Lolo National Forests, the Swan River State Forest, and Plum Creek Timber Company. The area is also becoming increasingly residential in character: The number of residential lots in the Swan Valley increased 30% between 1987 and 1993 (Missoula County Rural Planning Department, Lambrecht and Jackson 1993).

Along with a growing human population, the Seeley-Swan also is home to numerous wildlife species, from goshawks (*Accipiter gentilis*) to grizzly bears

(*Ursus arctos*). The central part of the valley is an important white-tailed deer winter range, and a USFWS refuge at the southern end of Swan Lake provides waterfowl habitat. Many sensitive plant species, including *Epipactis gigantea* and *Howellia aquatilis*, also are found in the Seeley-Swan.

In the digital world of GIS, the Seeley-Swan study area is represented by a set of data layers, as outlined in Table 2-1. Later chapters will expand upon construction of individual layers. All data are stored in Albers Conical Equal-Area projection, North American Datum 1927. For the most part, analyses were conducted using ARC/INFO 6.1.1 on an IBM RS/6000 workstation.

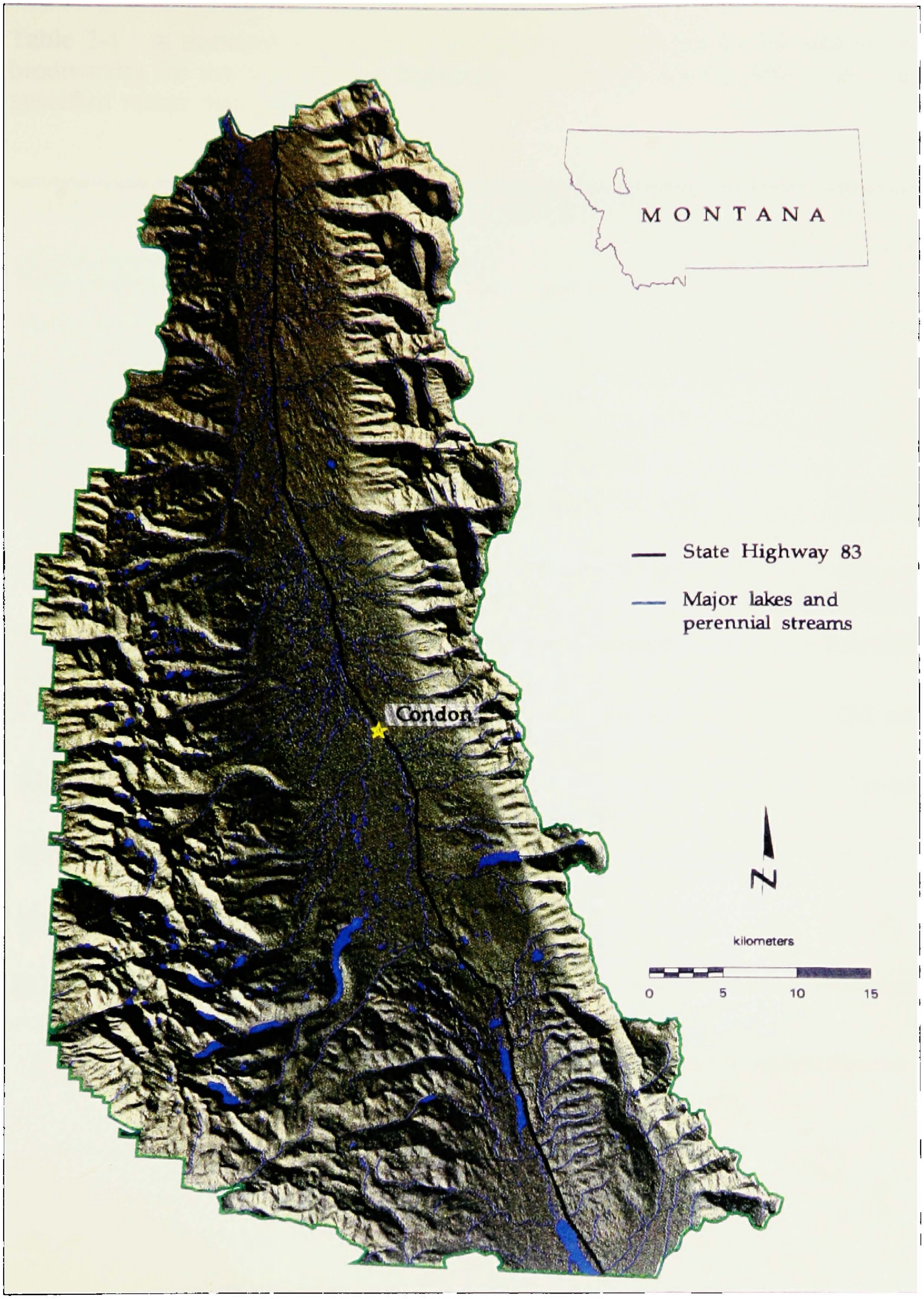


Figure 2-1. The Seeley-Swan landscape in northwestern Montana as portrayed by a digital elevation model. A broad valley separates the Mission Mountains to the west and Swan Range to the east.

Table 2-1. A summary of the basic GIS layers incorporated in the analysis of biodiversity for the Seeley-Swan landscape. For raster layers, cell resolution is specified rather than scale.

| DATA LAYER | TYPE | SOURCE[a] | SCALE (RESOLUTION) |
|---|---|---|---|
| Topography | raster | USGS 7.5 minute quads | (30 m) |
| Hydrography | vector | USGS 7.5 minute quads, digitized by Flathead NF and MTCWRU | 1:24000 |
| Ownership | vector | USGS 7.5 minute quads, digitized by MTCWRU | 1:24000 |
| Management | vector | composite (Flathead NF and MTCWRU) | 1:24000 |
| Roads | raster<br>vector | Flathead NF (from CFFs);<br>Lolo NF, CFFs | (10 m)<br>1:24000 |
| Sensitive species and unique habitats | vector | Montana Natural Heritage Program | 1:24000 |
| Current vegetation | raster | Landsat Thematic Mapper imagery | (30 m) |
| 1930s vegetation | vector | Forest Service stand maps digitized by MTCWRU | 1:31680 |
| Timber stands | raster | Flathead NF, Swan Lake Ranger District | (50 m) |
| Timber stands | vector | Montana Department of State Lands, Swan River State Forest | 1:24000 |

[a] USGS = U.S. Geological Survey; NF = National Forest; MTCWRU = Montana Cooperative Wildlife Research Unit; CFF = cartographic feature file.

## Chapter 3:
## Vegetative Patterns Across Temporal and Spatial Scales in the Seeley-Swan Landscape.

### INTRODUCTION

Landscapes, defined in an ecological sense, are kilometers-wide, heterogeneous areas made up of repeating clusters of interacting ecosystems (Forman and Godron 1986). As a management unit, the landscape is becoming increasingly popular; Noss (1983) suggested that it may be a more appropriate unit than individual sites or ecosystems, particularly in areas of high heterogeneity. This concept has gained wide acceptance in both research and management sectors, but formal methods for landscape assessment are in the embryonic stage. Only recently have such broad-scale assessments become feasible; the development of GIS permits quantitative assessment of ecological heterogeneity and its consequences over a wide range of spatial and temporal scales (Johnson 1990).

A critical element in landscape assessment is the evaluation of presettlement vegetation patterns and the processes of disturbance and succession that generated them; realistically, pattern and process cannot be separated (see Noss 1985). Evaluating presettlement vegetation is complicated by the problem of selecting a benchmark to define a landscape's natural condition: As Sprugel (1991) has asked, what is "natural" vegetation in an ever-changing environment? Noss (1985) described presettlement-type systems as relatively ancient and stable, offering a baseline for comparison with systems heavily influenced by humans, but noted that most evidence suggests that, over relatively brief time frames, complex ecosystems can be assembled by species behaving individualistically (sensu Gleason 1926). For example, Davis (1984) reported that forest communities in the northeastern United States show little long-term stability of species associations over time. Clearly, it is critical to avoid approaches involving attempts to force nature to conform to a landscape pattern observed at one point in time (Noss 1985); rather, attempts should be made to compare a landscape's current state to a range of natural conditions (Sprugel 1991). In assessments of presettlement vegetation, one should consult all possible sources and seek corroboration among them, incorporate qualitative and quantitative data, and preserve a healthy skepticism toward questionable sources (Noss 1985). In illustrating changes in the Seeley-Swan landscape over time, I have endeavored to follow the above advice. I must admit that my approach tends to provide snapshots for comparison, but I've tried to correct this by supplementing these snapshots with narrative descriptions of historic patterns and processes.

Obviously, to draw comparisons between historic and current vegetation patterns, base layers must be prepared for the periods of interest. A fairly detailed map of vegetation was constructed in the 1930s as part of a region-wide inventory of commercial forest lands (USDA:FS 1937-43), providing an excellent source of information for the period just prior to the onset of effective fire suppression (roughly 1940, Antos and Habeck 1981) and timber harvest activity. To assess landscape patterns after half a century of the aforementioned human activities, I assisted in preparing a map of existing vegetation through interpretation of Landsat TM-5 satellite imagery in conjunction with digital terrain data. Integration of remotely-sensed and ancillary data -- especially topographic variables -- has proven to be a fairly effective and accurate means of mapping large areas (Cibula and Nyquist 1987, Frank 1988, Bolstad and Lillesand 1992, Brown et al. 1993, Congalton et al. 1993).

To draw comparisons, I relied on landscape statistics calculated by the FRAGSTATS program (McGarigal and Marks 1994). I also analyzed the sensitivity of the resultant statistics to variations in map resolution, recognizing that changes in spatial scale can influence interpretations of landscape pattern (Turner et al. 1989). My overall objectives were to: 1) produce a map of existing vegetation (circa 1991) and assess its accuracy, 2) convert a map of historical vegetation (circa 1937) to digital format and assess its accuracy, 3) compare vegetation for the two time periods, with special emphasis on older forests, and 4) compare landscape statistics across minimum mapping units for existing vegetation.

## METHODS

## MAPPING HISTORIC VEGETATION

Historic vegetative patterns were assessed using maps prepared by the U.S. Forest Service. As part of a nationwide forest survey effort authorized by the McSweeney-McNary Forest Research Act of 1926, the U.S. Forest Service inventoried forested lands in western Montana and northern Idaho between 1932-43 (USDA:FS 1937-43.) In landscapes like the Seeley-Swan, where timber harvest was minimal and fire suppression ineffective prior to 1940, these inventories offer valuable baseline data for estimating vegetative changes over time.

### Original Methodology

A progress report issued by the Northern Rocky Mountain Forest and Range Experiment Station (USDA:FS 1937) provided insight into the Forest Survey methodology; key points gleaned from this report are outlined below. The inventory phase of the Forest Survey project concentrated on forest cover types, timber volume, and total forest area. Township maps showing forest and nonforest types were prepared at a scale of 1:31,680 (Fig. 3-1). "Salient features" of each forest type were recorded: type of stand, species represented, average size of the dominant trees, volume range, average age, stand density or stocking, site index, and harvest information (Table 1-1). In compiling these maps, available cruise and type data were first collected from public agencies and private owners. The reliability of existing data was then checked, and holes in information were filled to complete coverage of the region. Mappers were required to make extensive cruises of the sawtimber stands that had not yet been examined and to label immature and nonmerchantable types. Minimum mapping units were 40 acres (16 ha) for sawlog stands, and 100 acres (40 ha) for other forest lands; nonforest types were "mapped in some detail." The mappers delineated stand boundaries from vantage points across the landscape, then traversed each stand sufficiently to classify it. Intensity of coverage depended on the forest type; for example, a merchantable stand received more attention than a younger one. Maps were drafted in the field and later checked and completed in the office. There is no mention of the use of aerial photography in preparing the township maps, although photos from that period are known to exist, at least for much of the Seeley-Swan. Township maps for the Seeley-Swan were prepared by several mappers prior to 1940; most maps for the area are dated 1937. For convenience, they will be referred to throughout this paper as the 1930s data.

### Data Acquisition and Conversion to Digital Format

Maps were acquired from the archives of the Maureen and Mike Mansfield Library at the University of Montana. Maps for each township within the Seeley-Swan study area were digitized into ARC/INFO vector format, and attribute data were attached to each stand.

### Accuracy Assessment

Prior to digitizing, maps for the Placid Lake area were visually compared with 1934 aerial photographs to quickly assess accuracy of stand delineation. Recognizing that a cursory visual assessment was insufficient, a more quantitative method was later employed before extensive use was made of the 1930s data. Age estimates from the Forest Survey maps were compared with estimates taken from recent stand exams (Hart and Lesica 1994). Age was the only stand characteristic suitable for such a comparison. Evaluating the accuracy of cover type assignments might have been more desirable, but successional changes in stand composition precluded reliable comparisons between the two time periods.

## MAPPING EXISTING VEGETATION

Methods for mapping existing vegetation are especially important, because existing vegetation is the base layer upon which further analyses of biodiversity and recommendations for its protection are built. This map layer was constructed in conjunction with the Montana Gap Analysis project. Because the methodology has not yet been documented elsewhere, I will first outline the generic process before describing its application to the Landsat Thematic Mapper (TM) scene in which the Seeley-Swan landscape is located.

### Classification Process

A map of existing vegetation and land cover was developed by integrating multi-spectral Landsat TM data with ancillary biophysical data in a two-stage digital classification process (Ma and Redmond in prep.; Fig. 3-2). Superficially, the process resembles manual digitizing and labeling of polygons, but has the advantages of consistency, repeatability, and reduced processing time. The first stage is analogous to manually digitizing polygons, and the second to labeling cover types.

<u>First Stage: Classifying pixels</u>. The first stage is an unsupervised classification of pixels using a specially-designed algorithm, VISUALIZATION/ MAPPING, to identify spectral groups that simulate the enhanced false-color composite of TM channels 3, 4, and 5 (assigned to blue, red, and green). These channels are best for general cover type discrimination (Horler and Ahern 1986) because they generally have the least spectral overlap among cover types (Ma and Olson 1989), and may be best for discriminating species or age during periods of full foliage (Leprieur et al. 1988). In the classification program, the number of spectral groups is determined by two user-specified criteria: 1) the correlation between a pixel and a reference point in three-dimensional color space, and 2) the distance between a pixel and the origin (0,0,0) in color space (Fig. 3-3). The former seeks to match the color of each pixel with the false color composite,

whereas the latter emulates each pixel's brightness (the farther from the origin, the brighter). Pixels with similar color and brightness are assigned to the same spectral group.

In the first stage, the algorithm employs a two-pass process. The first pass searches for pixels that do not meet the above user-specified criteria, defining another spectral group whenever either condition is not met. It also randomly selects one pixel to represent each spectral group. With random selection, the probability of a group being selected is equal to the population size of the group, so none of the larger groups will be missed; if a small group is missed, it can be manually added later. These randomly-selected training pixels are stored and used in the second pass to classify all pixels into spectral groups. The resulting pixel classification closely resembles the false color composite in appearance.

After the second pass, classes are digitally regrouped with some modifications by the operator. Digital regrouping is an enormous timesaver, because conventional unsupervised classification methods require the operator to manually regroup classes. The classified image is then smoothed using a 3 x 3 window filter to remove "salt and pepper" regions ("raster polygons", or groups of pixels having the same value). This step improves the physical appearance of the image, and greatly reduces the number of records in the GIS database. The smoothed image then is subjected to a newly-developed merging process (MergeRP, a C++ program; Guo 1993, 1994). In the process, small regions are combined with highly similar, larger neighbors, thus creating regions equal to or larger than the desired minimum mapping unit. This process is distinctive in its reliance on a similarity matrix to determine which regions may be combined with one another: Other readily available methods (such as the NIBBLE function, ESRI 1991) merge small regions with neighbors regardless of attribute similarities. An approach based on similarities is much more realistic for natural resources applications, because highly dissimilar cover types (e.g., rock and forest) are unlikely to be combined.

Second Stage: Classifying Regions. The second stage of this process involves supervised classifications of regions rather than individual pixels. The spectral groups are converted into an ARC/INFO grid (raster layer, ESRI 1991), an attribute table is built, and mean values for the different spectral and biophysical variables (including the 7 TM channels, elevation, slope and aspect) are assigned to each region. Ground-truth data from existing vegetation maps, field plots, and aerial photographs are analyzed with spectral and biophysical data, and training sites are selected to represent the range of attributes to be mapped. Training sites for each attribute are then used in a series of supervised classifications through a nonparametric method called NEAREST MEMBER of GROUP (Ma, unpubl. software). The resultant image is labeled according to vegetation cover types, structural stages, and levels of canopy closure. Because the data are never converted from raster to vector format, many of the traditional limitations for building attribute tables are avoided, and large areas can be processed quickly and efficiently.

**Applying the Classification Process to a Full Scene**

The map layer of existing vegetation for the Seeley-Swan was derived from a Landsat-5 TM scene dated 20 July 1991 (WRS path 41 and row 27). This TM scene covers over 5 million ha of western Montana (Fig. 3-4). The scene was terrain-corrected by Hughes STX Corporation and rectified to Albers conical equal-area projection. An assessment of its planimetric accuracy showed average displacement from true positions (root mean square error) of 18 and 30 m in the x and y directions, respectively (Troutwine, unpubl. data).

Once obtained, the scene was classified in the two stages described above; steps in the process are illustrated in Figure 3-5. A digital layer with 70 spectral classes was produced; through regrouping, the number of classes was reduced to 31. Next, polygons were merged to 2 ha MMU. The merged layer was then cut into four pieces, and each was converted to ARC/INFO grid format. In each quarter scene, all regions were assigned unique identifiers. Next, for each region, mean values for TM bands 1-7, elevation, slope, and aspect were calculated and included as items in the grid file's attribute table.

Training sites were selected for use in the second stage, a supervised classification (or labeling) of regions. Training data were obtained from a variety of sources, including University of Montana (UM) plots, ECODATA plots (Keane et al. 1990), and stand databases for Swan Lake Ranger District and Swan River State Forest. University of Montana (UM) plots were collected in the summer of 1992, before the terrain-corrected Landsat image that was ultimately used had been acquired, and before the previously-outlined classification process had been fully developed. Locations of UM plots were selected to fill in gaps in coverage of the study area by ECODATA plots provided by Flathead National Forest; existing ECODATA plot locations were used to guide placement of UM plots in the following manner. On-screen, ECODATA plots were overlayed with a false color composite (bands 4, 5, and 3) of the portion of the Landsat scene within the study area, and photographs were taken of areas lacking plot coverage. Photographs were then converted to color Xerox prints to be used, in conjunction with 7.5' topographic quads and aerial photographs, in plot selection and field orientation. Several types of UM plots were used to characterize stands, ranging from quick walk-throughs to full ECODATA plots (USDA:FS 1992). On a weekly basis, UTM coordinates for the plots were determined using the CURSES module in ERDAS (1991), and all data were entered into an INFO database file to be later converted into an ARC/INFO point coverage.

ECODATA plots were obtained from the Flathead National Forest and the Intermountain Fire Sciences Laboratory. In addition, ECODATA plots were collected in 1993 by a Forest Service Region 1 field crew for use in this classification. Although we initially hoped to use a large number of existing ECODATA plots as training sites, relatively few were actually employed because of locational inaccuracies and difficulty of rapid interpretation.

The stand databases for Flathead National Forest and Swan River State Forest were also used as supplemental sources of training sites. For cover types poorly represented by the collection of useable UM and ECODATA plots, queries were written to generate new polygon coverages including only the individual cover types of interest. Polygon coverages were then overlayed with the grid layer and visually examined for polygon boundaries that corresponded well with the regions of the grid. Stand databases were a primary source of training sites for structure and canopy closure; sites were identified as described above for cover type. Nearly all training sites for cover types like water, snow, and rock were obtained from a combination of on-screen interpretation of the Landsat scene and examination of aerial photographs.

In all, 328 training sites were used to label regions according to cover type, 107 sites to label forest structure (size class), and 74 sites to label forest canopy closure. Training sites were examined for overlap in mean attribute values, which could create confusion in the supervised classifications. Three separate supervised classifications were conducted, one for each of the above characteristics (Table 3-2); this method was similar to one successfully employed by Congalton et al. (1993) in mapping old-growth forests in the Pacific Northwest.

After the supervised classifications were completed, a vegetation layer for the Seeley-Swan study area was clipped out of the GRID files for the full scene. Following this, some manual adjustments were made to the vegetation layer. To establish consistency between the three classifications, structural stage and canopy closure were set to 0 for all nonforest types, to 1 for all recent cuts and seedling stands, and 2 for all sapling stands. Several cover types, including agricultural lands, urban areas, and recent burns, were manually mapped because of 1) limited extent in the study area, and 2) difficulties in digitally distinguishing land use (as opposed to land cover) types in the first two cases. To identify agricultural lands, natural color aerial photographs (1:16,000) were examined for all privately owned parcels. All grasslands within the parcels were assumed to be agricultural unless human use (in the form of houses, roads, or cattle trails) was not evident. Corresponding regions in the vegetation layer were then identified and recoded. Urban areas were mapped in the same manner. Recently burned areas (within the last decade) were identified through contacts with Forest Service and Department of State Lands personnel, and then manually recoded.

## DERIVING STANDARD CODES FOR VEGETATION, 1930s AND 1990s

### Cover Type

In order to compare the historic and current Seeley-Swan landscapes, I drew parallels between the 1930s and 1990s vegetation maps by regrouping the codes. First, I built a list of standard codes for cover types (Table 3-3). I used definitions for 1930s and 1990s cover types to identify similarities. For 1930s commercial forest types, I examined data on species composition where recorded. I also created on-screen maps of each cover type for each time period, and displayed them side by side, searching for patterns to aid regrouping efforts. Some types were directly comparable between the two classification schemes, including water, agriculture, burn, urban, broadleaf, Douglas-fir, lodgepole, and spruce-fir types. The remainder required a number of assumptions; the logic behind my regrouping decisions is outlined in Appendix A.

### Size Class

I also standardized the codes for forest size class; here, the comparisons were relatively straightforward. Three standard size classes were derived: 1) recent cut/seedling/sapling, 2) pole/immature, and 3) mature/overmature. The first class corresponded to the recent cut/seedling/sapling standardized cover type. The pole/immature class included the 1930s pole stand class and the 1990s pole and immature size classes. The 1930s sawtimber stand class and the 1990s mature/overmature size classes were labeled mature/overmature.

**Stand Density**

Standard codes for density were not as simply derived; in fact, I was unable to draw a sound parallel between the two time periods. A number of terms with different shades of meaning are routinely used in forestry to describe stand density (Curtis 1970). In the 1930s, the closest measure of density mapped was stocking level. For seedling/sapling and pole stands, stocking levels were based on the percentage of area occupied by trees: Less than 10% utilized was considered unstocked; 10-40%, poorly stocked, 40-70%, moderately stocked; and 70-100%, well stocked (USDA:FS 1937). On the other hand, stocking levels for sawtimber stands were based on estimated volume in trees 14" DBH and larger (12" for *Pinus* species). For all forest types except ponderosa pine and lodgepole pine, stands with 4-10 mbf/ac were considered poorly stocked; 10-20 mbf/ac, moderately stocked; and >20 mbf/ac, well stocked. These limits were lowered for ponderosa pine and lodgepole pine to 3-7 mbf/ac, 7-13 mbf/ac, and >13 mbf/ac (USDA:FS 1937).

The measure of stand density mapped for the 1990s was canopy closure. As with stocking levels for the 1930s, canopy closure for the 1990s was mapped at 3 levels: low (≤29%), medium (30-59%), and high (≥60%). However, the two measures are not directly comparable, at least for sawtimber stands; it does not follow, for example, that a stand with low estimated volume necessarily has low canopy closure.

Attempting to draw some correlation between the two measures, I estimated crown competition factor (CCF, Krajicek et al. 1961), a measure of stand density based on the relationship of crown area to DBH for open-grown trees, in the following manner. For a given stand in the 1930s, I found the midpoint of the age range, the midpoint of the volume estimate, and the site class. These characteristics were used to look up trees per acre for the total stand (all diameter classes) and the DBH of the average tree in the stand. Next, I looked up the percentage of trees in each size class for the average DBH value, then multiplied the percentages by the total trees per acre to find trees per acre in each size class. CCF formulas were then applied for each diameter class (Wyckoff et al. 1982), using coefficients specific to the cover type of the stand and the DBH class, and the resultant CCF values were summed. Because these sums are CCF values for "normal", fully-stocked stands, the values had to be adjusted for stands with stocking levels below the norm. I looked up the volume figure predicted for the selected stand's age and site class and divided that figure by the volume estimate actually listed for the stand to obtain a percentage of normal. This percentage was then applied to the summed CCF value above.

Two assumptions are implicit in this method: 1) the stands were undisturbed, and 2) trees were not clumpily distributed (Krajicek et al. 1961). The first assumption is valid; sawtimber stands that had been selectively harvested were separately coded in the 1930s, and so I was able to separate these few stands from the undisturbed sawtimber stands and eliminate them from analysis. However, the second assumption is seldom completely satisfied in natural stands. Nevertheless, assuming stands have not been disturbed, a CCF value of 100 indicates complete canopy closure, and lesser values can be taken as approximations of canopy closure (Curtis 1970).

I was unable to calculate CCF values for all undisturbed sawtimber stands because volume tables and diameter distributions critical to this method have not been prepared for several species occupying much of the study area (e.g., western larch, Douglas-fir, Engelmann spruce, and lodgepole pine). Sufficient information was obtained to calculate CCF values for western white pine (Haig 1932) and ponderosa pine stands (Meyer 1938). Later, I will use these values to briefly illustrate how stand densities in the 1930s might compare to those in the 1990s, rather than include stand density in more formal comparisons of vegetative composition between the 1930s and 1990s.

## CALCULATING LANDSCAPE METRICS

### Comparing Historic and Current Landscapes

Once the vegetation coding had been standardized, I prepared the files for each time period for input to the FRAGSTATS program (McGarigal and Marks 1994) by converting them from ARC/INFO grid format to ERDAS GIS layers containing standardized codes combining cover type and size class data (Table 3-3). Because the minimum mapping units (MMUs) differed for the two time periods, I merged regions for the 1990s layer from 2 ha to 16 ha (the MMU for sawtimber stands in the 1930s) using the MergeRP program described earlier (Guo, unpubl. software); specific methods will be detailed in the following section. At this point, layers for the two time periods had the same vegetation coding and minimum mapping unit.

I then calculated statistics at the patch, cover type (or class), and landscape levels using the raster version of FRAGSTATS; it incorporates measures like nearest-neighbor distance which cannot be calculated in the vector version. Edge width was set at 60 m (2 pixels), patch richness at 25 types, and proximity search distance at 300 m (for consistency with Gustafson and Parker (1994)); defaults were accepted for the remainder of the parameters. Output files were analyzed with DataDesk for the Macintosh (Velleman 1994). Change in area between time periods was calculated. Because of positive skews in the patch-level data (from which higher-level measures are derived), Mann-Whitney U tests were calculated to compare a selected set of metrics for the two time periods. For each measure, the Mann-Whitney U tests evaluated whether the median value for all classes in the 1990s landscape differed from the median value for all classes in the 1930s landscape (both at 16 ha MMU).

### Examining Effects of Map Resolution on Landscape Interpretation

I quantified the effects of changing map resolution by comparing the structure of the 1991 Seeley-Swan landscape at eight MMUs: 2, 10, 16, 20, 40, 100, 200, and 400 ha. The 2 ha MMU is used by the Montana Gap Analysis project for data processing and storage. Polygons will be merged later to larger MMUs (40 ha for wetlands and 100 ha for upland types) for compatibility with Gap Analysis maps for other states. Several western states, including California, Colorado, and Wyoming, have manually digitized polygons at these larger MMUs, and are processing data only at this coarser scale. Through this method, large amounts of computer disk space and processing time are saved, but some degree of detail is sacrificed in the resultant landscape descriptions. My comparison is intended to clarify how these descriptions may change and information may be lost at coarser spatial resolutions.

The 2 ha vegetation layer for the Seeley-Swan was used as a baseline for comparison; layers for all other MMUs were constructed through manipulation of this layer. Files for each MMU were generated as follows: First, mean values were calculated by cover type for TM channels 1-7, elevation, aspect, and slope in the base vegetation (2 ha) file. Based on these mean values, a matrix of similarities between cover types was then calculated for later use. A new grid, with values corresponding to the standardized codes (Table 3-3), was next created from the base vegetation file, then converted to ERDAS GIS format (ERDAS 1991). The MergeRP program was then used to create a series of ERDAS GIS files for the desired MMUs; the similarity matrix was used by the program to determine which cover types should most logically be merged for regions smaller than the specified threshold value (which corresponded to the number of 30 m cells most closely matching the desired MMU in area). Water was excluded from the merging process -- for this type, polygons as small as 1 cell were preserved in the output layers for each MMU. However, water was later recoded to 0 (background) in ERDAS so that its smaller patches would not influence the calculation of landscape statistics.

To estimate sensitivity of landscape statistics to treatment of individual cover types (such as water) in the merging process, I compared three different treatments: 1) water was held to 2 ha MMU while other cover types were merged to 100 ha MMU and included in all calculations of landscape statistics, 2) water was held to a 2 ha MMU and then excluded from all calculations (treated as background), and 3) water was merged to 100 ha MMU along with all other types and included in all calculations.

All merged output images were processed using the raster version of FRAGSTATS, again with a 60 m edge width, patch richness of 25 types, and 300 m search distance for proximity analysis. Output files were analyzed with DataDesk. For a selected set of landscape measures, Spearman rank correlation with MMU was calculated.

## RESULTS

### Description of Vegetation

1930s. Thirty-four cover types were mapped for the Seeley-Swan landscape in the 1930s (Table 3-4). Dominant cover types included western larch/Douglas-fir, subalpine fir, lodgepole pine, and Engelmann spruce. Many cover types occurred at extremely low frequency. Sawtimber stands occupied the most area, and a majority of stands (in terms of area) were described as poorly or moderately stocked (Fig. 3-6). For stands where age information was recorded, roughly 48% of the total area was occupied by forests 200 years or older (Fig. 3-7). In particular, much of the valley floor was occupied by stands 200 years or older (Fig. 3-8).

1990s. Thirty cover types were mapped for the Seeley-Swan landscape in the 1990s (Table 3-5). Dominant cover types included mixed conifer, Engelmann spruce/subalpine fir, sapling, lodgepole pine, and whitebark pine/Engelmann spruce/subalpine fir. As with the 1930s data, many cover types occurred at low frequency. Mature/overmature stands occupied the most area, and most stands had moderate to high canopy closure (Fig. 3-9).

**Comparison of Vegetation -- 1930s versus 1990s**

Cover Type. Using the standardized vegetation codes (Table 3-3), comparisons of the two time periods were possible; Figures 3-10 through 3-12 depict cover type distributions. Table 3-6 shows changes in areal extent of cover types between the 1930s and 1990s. Fourteen of 25 cover types increased, most notably grass, shrub, seedling/sapling, pole mixed conifer, and mature/overmature Douglas-fir, lodgepole pine, and western red cedar. Overall, mature/overmature forest types declined (Figs. 3-13, 3-14). Several statistics are shown for each cover type in Table 3-7. The most noteworthy trend is the large variation both within and between periods for most metrics, including number of patches and mean patch size.

Stand density could not be included in this comparison for all cover types, but using the crown competition factor (CCF) equations, I tried to approximate canopy closure (as mapped for the 1990s) based on 1930s stocking levels for several types of western white pine and ponderosa pine stands (Table 3-8). Results support the hypothesis that low volume estimates do not always equate with low canopy closure for the 1930s sawtimber stands. Especially in the western white pine cover type, stands were likely composed of more than just a few, widely scattered trees. Of all cover types mapped, ponderosa pine is most likely to have a naturally open canopy; the calculated CCF values illustrate this nicely (Table 3-8).

Landscape. The number of patches in the Seeley-Swan landscape increased markedly between the 1930s and 1990s, whereas mean patch size decreased (Table 3-9). The coefficient of variance for mean patch size remained roughly the same (388% versus 396%) for the two time periods, however. Patch shapes became slightly more complex for the 1990s landscape, as measured by mean shape index and mean patch fractal dimension. Total edge increased dramatically, as would be expected given an increasing number of patches. Similarly, the mean core area index decreased, indicating that a smaller proportion of a patch could be counted as interior habitat in the 1990s than in the 1930s. Mean nearest-neighbor distance also decreased; patches of the same type are closer to one another in the 1990s than in the 1930s. Patch richness, or the number of different patch types in the landscape, was slightly higher in the 1990s than in the 1930s, but this is of little practical importance -- one of the two types found only in the 1990s landscape, cloud, is an artifact of the means of data acquisition; the other, pole-sized western red cedar, is at least partly a result of misclassification (see Discussion). The diversity and evenness indices are fairly similar for both landscapes, as are the measures of interspersion/juxtaposition and contagion. A combination of relatively high values for interspersion/juxtaposition and low values for contagion suggests that cover types are fairly well interspersed in both landscapes. Mann-Whitney U tests revealed statistically significant differences ($p \leq 0.05$) over time for three measures: mean shape index, mean patch fractal dimension, and number of core areas. However, practical differences are apparent for other metrics, including number of patches, mean patch size, and total edge. In such instances, large variation between cover types within time periods may have precluded the detection of differences between time periods.

**Comparison Across Scales**

Treatment of Water. Naturally, the method by which any single cover type, including water, is treated in landscape analyses will influence the results. A comparison of three treatments is presented in Table 3-10: 1) water is held to a 2 ha MMU while other cover types are merged to 100 ha MMU, then included in calculation of landscape statistics; 2) water is held to a 2 ha MMU and then excluded from all calculations (treated as background); and 3) water is merged to 100 ha MMU along with all other classes and included in all calculations. The most striking differences are seen in the number of patches and mean patch size; other differences are primarily a function of these two measures. Inclusion of water at a 2 ha MMU makes the landscape appear much more fragmented, as shown by the mean core area indices. The diversity and evenness indices are quite similar between all treatments; this might be interpreted as robustness, or conversely as an inability to distinguish between landscapes. In general, landscape statistics were similar for treatments 2 and 3; I opted to treat water as a background class in subsequent analyses because that method best accords with Montana Gap Analysis methodology.

Proportion of Cover Types. As MMU increases, the proportion of the Seeley-Swan landscape occupied by each cover type changes (Fig. 15a-c, Table 3-11). Six of the 25 cover types, all forested, increase in areal extent, while 10 cover types disappear completely. As a group, nonconifer cover types decreased in areal extent (Fig. 3-16). Seventeen cover types increased or decreased monotonically, while the remainder fluctuated slightly as MMU increased.

Because polygon size is assumed to play the dominant role in the process of merging polygons, the distribution of polygons by size class was examined for the base vegetation layer (Fig. 3-17). Note that although 24,903 polygons were created (based on spectral group codes) and maintained in the ARC/INFO database for this layer, there are 13,247 polygons when recombinations are made based on standardized codes (Table 3-3). Nearly 70% of the polygons in the 2 ha layer were smaller than 10 ha. Table 3-12 breaks down this distribution by cover type: The most notable trend is that all but one of the cover types that showed an overall increase initially had polygons larger than 400 ha, ensuring their persistence at the 400 ha MMU and also enabling them to absorb the area of surrounding polygons smaller than the MMU. Obviously, cover types lacking individual polygons larger than the specified MMU should be less likely to persist in the landscape, yet DF2, PP2, and RC1 do so, perhaps because neighboring polygons had the same cover types and thus effectively met the required MMU. The initial distribution of cover types within the landscape also appeared to influence persistence; small fragmented types were more likely to be lost, while large contiguous types tended to increase.

Landscape Statistics. A selected set of landscape statistics was examined at increasing MMUs (Table 3-13); Spearman rank correlations exhibit almost perfect correspondence between most statistics and MMU. Statistics did not prove to be stable as MMU increased; rather, they tended to increase or decrease (Table 3-14). Furthermore, most changes were monotonic, yet relationships between MMU and individual statistics were distinctly nonlinear (Fig. 3-18). Thus, care must be used in interpreting statistics calculated at different resolutions, because the landscape portrait will change as MMU increases. With increasing MMU, the landscape appears to be composed of fewer, larger patches with more complex shapes. Total edge decreases and the amount of interior habitat (mean core area index) shows a corresponding increase. Distance between nearest neighbors increases, which can be attributed mostly to increasing patch size. Patch richness declines sharply as those cover types that are rare or typically occur in small patches are eliminated from the landscape. The diversity and evenness measures remain relatively constant, either due to actual robustness or inability to detect change. I suspect the latter is a more accurate representation; in a landscape where 10 out of 25 cover types disappeared between 2 and 400 ha MMU, one might expect wider variation in these indices. Contagion increases as patches become larger and more contiguous, and the index of interspersion/juxtaposition decreases.

## DISCUSSION

### Accuracy of Vegetation Layers

1930s. Hart and Lesica (1994) found that 52% of the 1930s stand age estimates were within 20 years of estimates taken in recent stand exams; no significant bias toward either higher or lower estimates was noted. They concluded that the 1930s Forest Survey data could be used with only limited confidence for individual stands, but should provide a reasonably accurate estimate of stand-age distributions over large areas. Extending this conclusion, I assumed that Forest Survey data on cover type, stand class, and stocking could also be appropriately applied to assessments of vegetative patterns in the Seeley-Swan landscape.

Maps for the 1930s were drafted from field surveys, not aerial photographs; thus, boundaries are likely to be generalized and inaccurate. For example, section lines were used to separate one stand of ponderosa pine from surrounding western larch/Douglas-fir; both stands were undisturbed, yet straight lines were used to delineate their boundaries -- an ecologically improbable separation. In addition, efforts were driven by the goal of mapping commercial timber, so less attention was paid to noncommercial types and higher elevations; instead, merchantable species and accessible areas were targeted. Finally, as a result of the 16 ha MMU, types usually occurring in smaller patches are likely to be underrepresented. Examples include wet meadows, riparian stringers (including western red cedar), and remnant patches of older trees within larger burned areas. Despite these limitations, the 1930s maps provide valuable information about historic vegetation patterns; although inaccuracies may be found for individual stands, patterns for an entire landscape should provide a fair representation of that period (Hart and Lesica 1994).

1990s. The vegetation layer for the 1990s was a draft dated December 1993, and was prepared as a pilot study for vegetation mapping across Montana; it has not been subjected to a formal assessment of accuracy. It offered the best available information at the time, but has known limitations. First, the number of training sites was limited, especially for less-common cover types. Furthermore, the diversity of sources used to identify those sites probably yielded inconsistencies within the training data set, which in turn may have generated errors in the supervised classification process. Such errors will be discussed in conjunction with temporal comparisons of vegetation.

**Suggestions for Improving Classification Methodology**

Obviously, I have no control over the methodology employed in the 1930s, and can only hope to illuminate potential problems with applying the results to landscape interpretations. However, if I were to attempt to classify existing vegetation based on satellite imagery again, I would follow a modified procedure:

1) Conduct an unsupervised classification of the TM scene.
2) Merge the classified image to a 2 ha MMU.
3) Create an ARC/INFO grid for use as a base layer.
4) Begin building training set:
   a) Identify types from aerial photos where possible.
   b) Groundtruth types that cannot be identified from aerial photos:
      - Find areas on the scene that are highly diverse in terms of spectral types or that contain rare types.
      - Print 1:24,000 color maps of these areas for use in the field.
      - Survey polygons, record basic information on desired vegetative characteristics (cover type, size class, and canopy closure), and mark locations on maps. Obtain differentially-corrected global positioning systems (GPS) locations for plots whenever possible.
      - Use existing data sources where necessary to increase the number of training sites, but reserve these sources primarily for accuracy assessment.
5) Enter all training sites into a database file, then conduct supervised classifications for desired vegetative characteristics.
6) Assess accuracy of supervised classifications.

I encountered problems because I collected field data before the first three steps had been completed. The TM data had not yet been classified, and I was guided in the field by fuzzy, color Xerox prints of the false color composite (channels 4,5,3) as displayed on-screen. My plot locations were tied to this image, which was found to be poorly georeferenced. We corrected this problem using a number of control points for the Seeley-Swan, conducted an unsupervised classification, and used the NIBBLE function (ESRI 1991) to merge polygons to a 2 ha MMU. I then relocated my plots in relation to this newly-created layer, and found many plots to be unusable because of changes in shape between the false color composite and the nibbled layer. In addition, I used existing data sources to increase the size of the training set, again based on the nibbled layer -- this was a laborious process, requiring intensive hands-on display and query of data layers.

In July of 1993, this work was jettisoned. A new, terrain-corrected scene was acquired, classified, and merged to a 2 ha MMU using the newly-developed MergeRP program described earlier (Guo, unpubl. software). Thus, an entirely new base layer was created. This new layer exhibited limited correspondence with the nibbled layer to which all training sites were registered, but better preserved the shapes seen on the false color composite. Switching base layers necessitated numerous adjustments to the training set. My primary reason for reciting this history is to highlight the importance of constructing a base layer before collecting training sites and evaluating their locations with regard to specific polygons. Violating this seemingly intuitive (in retrospect) principle cost me much effort, and left little time for evaluating and improving classification results.

**Comparing Vegetation, Past and Present**

Cover Type. Observed differences for individual cover types between the 1930s and 1990s may be attributed to actual alterations in vegetative patterns, or artifacts of differences in classification methods and accuracy. For most cover types, both factors are likely to be implicated. Water covers approximately the same amount of area in both time periods; the slight increase over time is probably observed because, in the 1990s, individual 30 m cells of water were maintained and some areas of snowmelt and runoff were classified as water. The cloud type was, of course, mapped only for the 1990s, and represents nothing more than a percentage of the study area that could not be mapped due to obstructions.

Barren, rocky woodland, and whitebark pine cover types all decreased substantially; I suspect these differences are related to parallel increases in grass, shrub, and seedling/sapling types. At higher elevations, very open forests may be classified as grass, shrub, or seedling/sapling because reflectance values are dominated by the understory, while the sparse overstory remains undetected. Similarly, areas of high whitebark pine mortality since the 1930s may now be classified as grass or shrub; in western Montana, whitebark pine has experienced high mortality rates over the past 20 years, primarily due to infection by white pine blister rust and epidemics of mountain pine beetle (Keane and Arno 1993). Increases in grass and shrub types can also be ascribed to misclassifications of harvested areas; because many young plantations are grass- and shrub-dominated, their reflectance values are similar to natural meadows and shrub fields.

A number of factors may be involved in the increased area occupied by seedling/sapling stands. First, such increases are at least partly due to timber harvest activities, which began in earnest a decade or so after the 1930s mapping efforts. The vast majority of seedling/sapling stands in the 1930s were initiated after fires; very few areas had been harvested at this time, and were mostly restricted to the periphery of the landscape. By the 1990s, this pattern had been reversed as a result of fire suppression and increased harvest activity. However, it appears that some open areas at high elevations were classified as seedling/sapling stands in the 1990s, but cannot be attributed to timber harvest. Some of these are probably misclassifications of meadows, rocky slopes, or open whitebark pine stands as described above. An additional factor relates to the 16 ha MMU I chose for comparison of the two time periods; in the 1930s, a 40 ha MMU was used for seedling/sapling and pole stands. Applying a 16 ha MMU in the 1990s would have allowed some stands that would not have been mapped by 1930s standards to remain in the landscape, and thus artificially increased the areal extent of seedling/sapling stands. Yet a similar or even more extreme contrast may have been detected if I had chosen to compare the two time periods at 100 ha MMU, because seedling/sapling is one of the cover types that increases in areal extent as MMU increases (Table 3-12). Regardless of the reasons for its increase, seedling/sapling stands have become more dominant over time, and now vie with mature/overmature forests for status as the matrix component of the Seeley-Swan landscape.

Broadleaf types were not prevalent in either time period, and disappeared in the process of merging the 1990s vegetation layer to a 16 ha MMU. Although hardwood species are a significant component of many forests in the Seeley-Swan today, pure stands of cottonwood, aspen, or paper birch are rare and tend to be quite small, rendering them less likely to be mapped. The mixed conifer and shrub types probably include some proportion of broadleaf species.

Urban and agricultural areas are slightly more extensive, representing increased settlement in the Seeley-Swan, but also preservation of its strongly rural character. However, residences are much more widely dispersed throughout the valleys today (although I made no effort to document this trend), and undoubtedly exert a heavy influence on landscape function, particularly with regard to wildlife movements. Human influence is also implicated in the striking decline in recently burned areas over time. Recent burns were still well-distributed throughout the 1930s landscape, but by the 1990s they were nearly absent, reflecting the effectiveness of fire suppression in the last half century.

Overall, pole stands increased slightly (about 7%) in total area, while mature/overmature stands decreased by 22%. Mixed conifer pole stands more than doubled. Because the mixed conifer type served as a catch-all class (mixed species composition is far closer to the rule than the exception in the Seeley-Swan), it is difficult to pinpoint the reasons for this increase. However, it is plausible that a broad range of attributes in the training set for this type led to various misclassifications. On the other hand, mature/overmature mixed conifer stands were halved, most likely because of timber harvest in the lower elevations where this class tends to occur, although some degree of classification error is undoubtedly involved as well.

Douglas-fir pole stands decreased; the increase in mature/overmature Douglas-fir logically accounts for this trend. The increase in mature stands also may be partly a side-effect of fire suppression: Some ponderosa pine-dominated stands may have filled in with Douglas-fir in the absence of frequent ground fires.

The amount of ponderosa pine pole stands increased slightly, but a sharp decline was observed for mature/overmature stands. As a valuable commercial species located in the accessible valley bottoms, ponderosa pine was an obvious target for harvest. Also, as described above, ponderosa pine stands are more likely to have a heavy component of Douglas-fir after 50 years of fire suppression, and may have been classified as Douglas-fir.

Pole stands of lodgepole pine decreased, and mature/overmature stands increased in a natural transition between size classes. However, it must be noted that the mature/overmature lodgepole pine class is in one sense a misnomer -- few stands of lodgepole are likely to meet the 14" DBH cutoff for this size class. Their inclusion in this category is probably a function of high stand density, which has a more obvious influence on reflectance values than actual tree sizes. Nonetheless, I feel these stands can still be appropriately termed mature/ overmature examples of lodgepole pine. It is also worth noting the proportion of lodgepole pine in the total area of mature/overmature stands for the 1930s (0.0007) and 1990s (0.2285). If not for this increase, an overall decline of much greater magnitude would have been observed for mature/overmature forests over time in the Seeley-Swan.

The observed increases in western red cedar types are probably an artifact of undermapping in the 1930s and overmapping in the 1990s. At the 16 ha MMU used in the 1930s, many cedar stands would remain unmapped. Obtaining sufficient training sites for cedar was difficult; perhaps some of the training sites used in the vegetation classification were poor representatives of pure cedar stands, and thus broadened the range of spectral values for the cedar type, causing more polygons to be classified as that type.

Reasons for increases in area of pole-sized Engelmann spruce/subalpine fir are unclear, but certainly include the combination of successional changes and classification errors hypothesized for other types. Mature/overmature stands decreased, but not as sharply as the lower-elevation mixed conifer and ponderosa pine types, where logging has been more extensive.

Overall, this comparison of vegetation is limited, of course, by the classification schemes employed. Riparian and wetland types were not adequately mapped for either time period, and thus no comparisons were drawn. However, these are important landscape elements widely distributed throughout the study area, and their inclusion in classification schemes would be a decided improvement. Furthermore, the classifications fail to account for the patchy nature of individual stands. Within many harvested areas, remaining trees create a complex mosaic adding some structural diversity to the landscape. A classification scheme which captured this textural aspect would add a further dimension to landscape characterization.

Landscape. The major differences in the landscapes for the two time periods -- number of patches, mean patch size, and mean core area index -- suggest a more fragmented landscape in the 1990s than existed in the 1930s. Coupled with these structural differences are differences in composition, or the proportion of the landscape occupied by each cover type. Such differences are likely to have important implications for landscape function.

Before discussing those implications, it would be useful to describe the patterns and processes assumed to dominate the Seeley-Swan landscape prior to European settlement based on a turn-of-the-century account. H.B. Ayres (1900) surveyed resources in the Lewis and Clark Forest Reserve, including the study area (Fig. 3-19, 3-20), for the U.S. Geological Survey. Although his maps were constructed at a scale too coarse (1:500,000) to be incorporated in a comparison with current vegetation, his report provides an invaluable historic portrait of the landscape. In 1899, Ayres viewed a landscape dominated by fire: About one-third of the Seeley-Swan had burned in the last 40 years. Ayres blamed most of the fires on human carelessness (settlers, hunters, prospectors, and Native Americans), though he noted that some were undoubtedly caused by lightning. Although Ayres blamed the extensive fires he saw upon humans, fire has long been recognized for its strong influence on forests in the northern Rocky Mountains over at least the past several hundred years (see Arno 1980). To some extent, especially in the lower-elevation forests in and near western Montana's major valleys, fire occurrence was increased by Native American ignitions (Barrett and Arno 1982). Such impacts cannot be considered any more natural than those caused by European settlers (Noss 1985); thus, one more complication is layered upon the already difficult task of assessing presettlement conditions.

Ayres (1900) reported that most of the fires had occurred in 1889, an exceptionally dry year, or around 1860. Although most of the burns he mapped were of stand-replacement intensity, many less intense fires had also crept over wide areas. The upper half of the Swan Valley had been extensively burned, and was blanketed by fallen trees. In this area, fires were moderate, thinning the forest. The lower Swan also was scarred by fires, but it had a great deal of older mixed forest; species typical of mesic sites were found in this region, including western red cedar, western white pine, and western hemlock. Ponderosa pine forests around the Swan headwaters and in the Clearwater drainage were subjected to repeated fires, and generally had more open understories than stands dominated by other species. Probably many western larch stands had similar fire regimes: Koch (1945) described a stand of 4-7' DBH larch on the west side of Seeley Lake as open, park-like, and sunny.

Although Ayres (1900) implicated humans in fire ignitions, their impacts on the landscape were otherwise fairly limited. Several squatters were located at the head of Swan Lake; between there and Holland Ranch, only ten unoccupied cabins were found. One cabin was also noted at the head of what is now Seeley Lake, but settlement was not widespread at this time. Trees had not been harvested for commercial purposes within the reserve, but logging operations were underway just to the south. Also, forest rangers had actively suppressed a few fires during the year of Ayres' survey.

Ayres' (1900) description accords well with the landscape mapped in the 1930s. The older forests he noted in the north end of the Swan Valley are evident in the 1930s maps as well, and the widespread seedling and sapling stands shown in the south end of the valley on 1930s maps bear witness to a fire history similar to that reported by Ayres. Certainly, some changes had occurred during the decades separating the reports: Presumably settlement increased, and some areas were harvested and others prevented from burning by diligent rangers. Landscape function had not changed markedly, but it was in a transition phase between domination by fire and domination by man. Over the next half century, this transition was completed, as fire suppression became effective around 1940 (Antos and Habeck 1981), and timber harvest became prevalent after 1960 (USDA:FS 1994a).

As a result, the managed landscape of the 1990s exhibits different patterns than the more natural 1930s landscape, including smaller and more numerous patches with more edge and less interior habitat. Individual stands have become more dense and fuels have accumulated as fires have been suppressed. Along with the harvest of timber and the building of residences, the road network has been extensively developed. All three activities modify habitat, create barriers for some species and conduits for others (most notably exotics), and increase the probability of human-wildlife conflicts. They also increase the likelihood of fire ignitions by humans (Franklin and Forman 1987). Fires may be more intense due to changes in stand composition; the low-intensity ground fires of the past are less likely to occur in today's landscape than are stand-replacement events.

In addition, the landscape matrix is less clearly defined today than it was in the 1930s. According to Forman and Godron (1986), the landscape matrix occupies more area, exhibits greater connectivity, and exerts greater control over landscape dynamics than any other type present. By the first two criteria, mature/overmature forests most likely functioned as a matrix in the 1930s (Figs. 3-13, 3-14). However, by the 1990s, seedling/sapling stands had increased greatly in area and showed great connectivity, at least in the valley bottoms. Mature/overmature forests maintain their role as landscape matrix, but doubtless the control they exercise over landscape dynamics has been weakened.

Not only has the amount of matrix diminished, its character has also changed. Fire suppression has allowed older stands to become denser and accumulate more standing and downed woody material. Habeck (1988) stated that, for western Montana, the term old growth often is applied to late seral, mature subclimax forests 200-500 years old, maintained originally in an open-canopied savanna state by frequent, low-intensity ground fires. He further suggested that parklike ponderosa pine and western larch stands may not qualify as old growth in today's context, lacking sufficient dead snags and decadent elements. However, upon examination of the current old growth definitions used by the Forest Service (Green et al. 1992), it is clear that attempts have been made to recognize the naturally open tendencies of these forest types. It is equally clear that the mature/overmature forest size class I have used for comparison between the 1930s and 1990s is not directly equivalent to old growth.

Estimating the amount of old-growth forest in the 1930s requires a number of assumptions about the attributes mapped in that period. I relied on stand age class (ranging from 0-20 to >200 years), stand class (seedling/sapling, pole, and sawtimber), and stocking levels. Sawtimber stands were labeled mature/ overmature for comparison with 1990s vegetation; the distribution of ages within the sawtimber class is heavily skewed toward stands >200 years old (Fig. 3-21). Minimum age criteria for all old-growth types in western Montana except lodgepole pine are 170-180 years (Green et al. 1992); all sawtimber stands in the 160-200 year and >200 year age brackets are shown in Figure 3-22. As noted earlier, stocking levels were low for most stands in the Seeley-Swan; however, my attempts to reconstruct stand densities suggest that despite low volume estimates, stands were composed of more than just a very few scattered large trees. I thus assumed that stand stocking should not be a limiting factor, and based my determinations on stand age and designation as sawtimber. Equating old age with old growth is a common error (Habeck 1988) and may lead to overestimation of old growth, but is a fairly sound approach based on limited information. I am unable to compare old growth between the two time periods because of limitations in the 1990s classification. Given an observed decline in mature/overmature forests over time, it is reasonable to assume that a similar trend would be seen for old growth.

**Comparing Vegetation Across Scales**

There is no one inherent scale at which ecological systems should be examined (Levin 1992). To better understand ecological heterogeneity, studies should be conducted across a range of scales and parameters robust to changes in scale should be identified (Milne 1991). It may be possible to predict or correct for information lost with changes in spatial scale by characterizing the relationship between ecological measurements and grain or extent (Turner et al. 1989). *Grain*, or resolution of data, and *extent*, or overall size of a study area, are both incorporated in definitions of spatial scale (Turner 1990); I focused on the former aspect.

My results suggest that care must be taken in interpreting landscape statistics calculated at different resolutions, because the landscape portrait will change as MMU increases. Most obviously, the relative proportion of cover types will change as smaller patches disappear. Rare cover types also may be lost with increasing MMU, causing the landscape to appear less diverse and potentially eliminating valuable elements from further consideration in evaluations of biodiversity. In comparing the distribution of coastal sage scrub in southern California at 1 and 100 ha MMUs, Stine et al. (1993) found that finer-grained representations do not necessarily nest within coarser-grained ones. They noted that if coarser maps are used to guide detailed local studies, small remnants -- which might serve as refugia and provide connections between larger reserves -- may never be considered. As habitat maps were generalized, Stoms (1992) found that some habitat types were locally eliminated, and thus the number of predicted species decreased. Turner et al. (1989) also observed that rare cover types were lost at coarser resolutions; dispersed cover types were rapidly lost, while clumped cover types were retained or slowly dwindled. Similarly, Moody and Woodcock (1994) found that cover types made up of large, homogeneous patches grew larger as resolution was degraded. Cover types with highly clumped distributions but smaller patches first grew, then diminished as the resolution exceeded that of most patches within the type. Small, fragmented cover types disappeared quickly in the aggregation process. I observed similar trends as MMU increased. Moody and Woodcock (1994) concluded that proportional errors become evident as land-cover data are sampled at progressively coarser scales, and may be significant enough to compromise the utility of the maps produced. This has important implications for Gap Analysis projects operating at a fairly coarse resolution (100 ha). The approach taken by the Montana Gap Analysis project may, however, alleviate this potential problem: Land cover types are determined at 2 ha MMU and polygons are then merged to 100 ha MMU. Information can thus be retained about the relative proportion of cover types within polygons, and rough locations for rare cover types will still be available.

Relative proportion of cover types in a landscape is not the only characteristic to shift as resolution grows coarser; other commonly-calculated landscape statistics also exhibit scale-dependent behavior (Turner et al. 1989, Lehmkuhl and Raphael 1993). Definite nonlinear relationships were apparent between MMU and most statistics I interpreted, suggesting that equations may be fit to these curves and extrapolations made between scales. Without such extrapolations, extreme caution should be exercised in drawing comparisons between landscapes mapped at different resolutions. Although most landscape statistics do not exhibit stability over a range of resolutions, this exercise suggested that their values may change in a predictable manner. The relationships behind these changes remain poorly understood; further exploration offers better understanding of the behavior of individual indices. More importantly, however, by establishing methods for extrapolation of results between landscapes, we stand to gain improved knowledge of ecological heterogeneity itself.

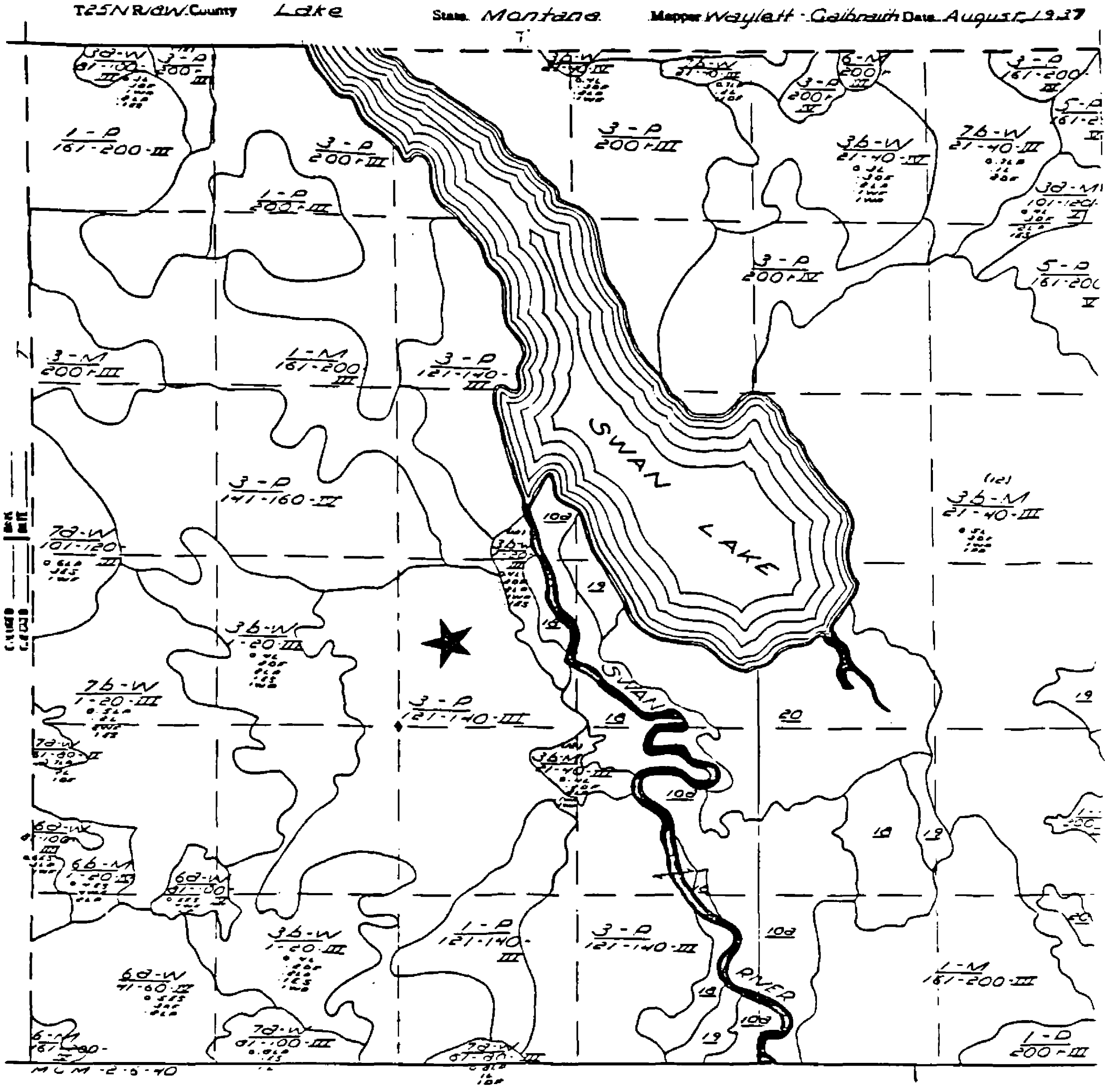


Figure 3-1. A sample township map prepared in the 1930s for the Swan Valley. Stand codes are listed in Table 3-1. For the starred stand, 3 indicates a western larch/Douglas-fir cover type, poorly stocked (P), 121-140 years old, with a site class of III.

Table 3-1. Legend for 1930s vegetation maps, compiled from USFS (1937), USFS (1937-1940), and USFS (1935). Much of the legend was reconstructed by B. John Losensky (pers. comm.). Not all types are found within the Seeley-Swan study area.

**COVER TYPE CODES**

"...based primarily upon existing characteristics or preponderating commercial species, with volume the index of preponderance." (USFS 1935:12,13).

| | |
|---|---|
| 1 | Western white pine (≥15% stand volume) |
| 2 | Mixed Douglas-fir/ponderosa pine (≥25% stand volume in pine) |
| 2.8 | Pure ponderosa pine (≥80% stand volume) |
| 3 | Western larch/Douglas-fir (≥75% stand volume) |
| 4 | Western hemlock/grand fir (≥50% stand volume) |
| 5 | Douglas-fir (≥60% stand volume; ≤10% western larch) |
| 6 | Engelmann spruce (≥50% stand volume) |
| 7 | Lodgepole pine (≥50% stand volume) |
| 8 | Western red cedar |
| 9 | Western red cedar/grand fir |
| 10 | Cottonwood (primarily river bottoms) |
| 11 | Subalpine (stands at the upper limit of tree growth, usually unmerchantable due to poor form and small size; may include subalpine fir, Engelmann spruce, alpine larch, whitebark pine, lodgepole pine, and mountain hemlock) |
| 12 | Restocked cutover area (pre-1925) |
| 12x | Selectively logged areas (remaining volume insufficient to classify stand as sawtimber) |
| 13 | Non-restocked cutover commercial area |
| 14 | Recent nonstocked cutover areas (post-1925) |
| 15 | Commercial, non-stocked burn (pre-1925) |
| 16 | commercial, non-stocked recently burned (post-1925) |
| 17 | Barren (too rocky, scanty as to soil, or exposed to support vegetative cover of trees, shrubs, or herbs) |
| 18 | Grass (parks, mountain meadows, treeless ridges) |
| 19 | Brush (sagebrush, brush, or shrubs -- a permanent type) |
| 20 | Cultivated (cleared and/or cultivated for agricultural use, including pasture) |
| 21 | Stump pasture (logged or burned land, part of operating farm units, mainly devoted to grazing; stumps or snags not removed) |
| 22 | Juniper (greater than or equal to 80% stand composition) |

Table 3-1 continued. Legend for 1930s vegetation maps.

**COVER TYPE CODES continued**

| | |
|---|---|
| 23 | Rocky noncommercial (within the range of commercial timber, below the limits of the subalpine type, and too rocky, steep, or sterile to produce a stand of commercial size, density, and quality; does not include areas economically inaccessible at that time) |
| 24 | Water (code added for use in ARC/INFO database) |
| X | Cutover (typically used in association with species codes) |
| W | Woodland (also used in association with species codes) |

(Codes may be combined, particularly for nonstocked, burned, or noncommercial areas; for example, 23-5 indicates rocky, noncommercial Douglas-fir.)

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STAND CLASS CODES

| | |
|---|---|
| (blank) | Sawtimber stands (majority of volume in trees ≥12" DBH for WWP, PP, LP; ≥14" DBH for all others) |
| a | Pole stands (majority of dominant trees 6-12" or 6-14" DBH, depending upon cover type) |
| b | Seedling and sapling stands (most of the dominant trees <6" DBH) |
| x | Cutover stands |

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STOCKING LEVELS

Sawtimber stands:

| | |
|---|---|
| P | Poorly stocked, 4-10 mbf/acre (PP and LP: 3-7 mbf/acre) |
| M | Moderately stocked, 10-20 mbf/acre (PP and LP: 7-13 mbf/acre) |
| W | Well stocked, >20 mbf/acre (PP and LP: >13 mbf/acre) |

(Areas <3 or 4 mbf/acre typed as immature or as one of the restocking or deforested types.)

Seedling/sapling and pole stands:

| | |
|---|---|
| U | Unstocked, <10% area occupied by trees |
| P | Poorly stocked, 10-40% area occupied by trees |
| M | Moderately stocked, 40-70% area occupied by trees |
| W | Well stocked, 70-100% area occupied by trees |

Table 3-1 continued. Legend for 1930s vegetation maps.

**SITE CLASS CODES**

Western white pine, western hemlock, grand fir, western red cedar, western red cedar/grand fir (Haig 1932).

| CLASS | SITE |
|---|---|
| I | 70 |
| II | 60 |
| III | 50 |
| IV | 40 |
| V | 30 |

Ponderosa pine (Meyer 1938).

| | |
|---|---|
| I | 127 |
| II | 112 |
| III | 94 |
| IV | 77 |
| V | 64 |
| VI | 50 |

Western larch/Douglas-fir, Douglas-fir (Cummings 1937).

| | |
|---|---|
| I | 75 |
| II | 65 |
| III | 55 |
| IV | 45 |
| V | 35 |
| VI | 25 |

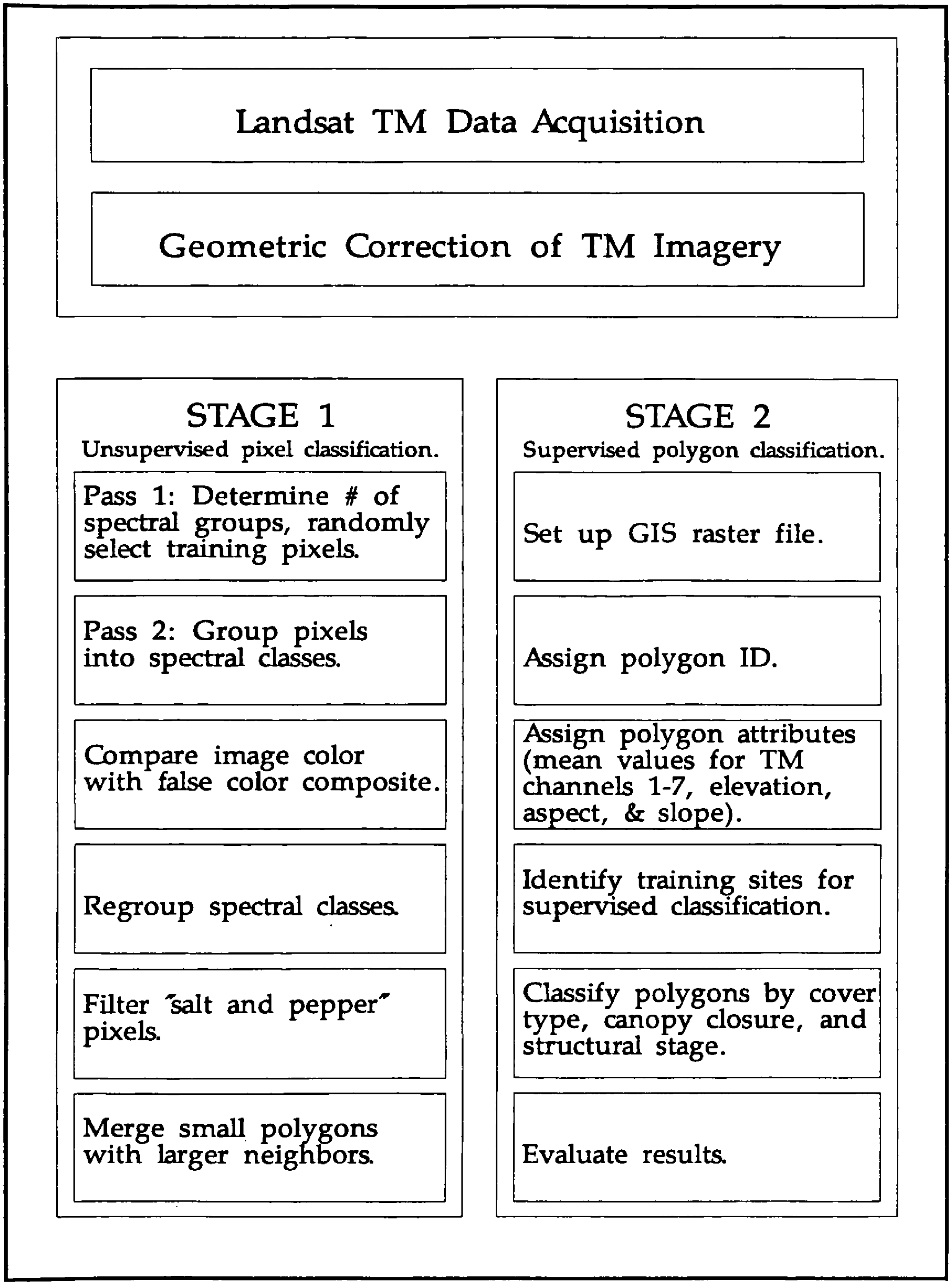


Figure 3-2. Schematic of two-stage process developed by Montana Gap Analysis project for classifying existing vegetation from Landsat TM data.

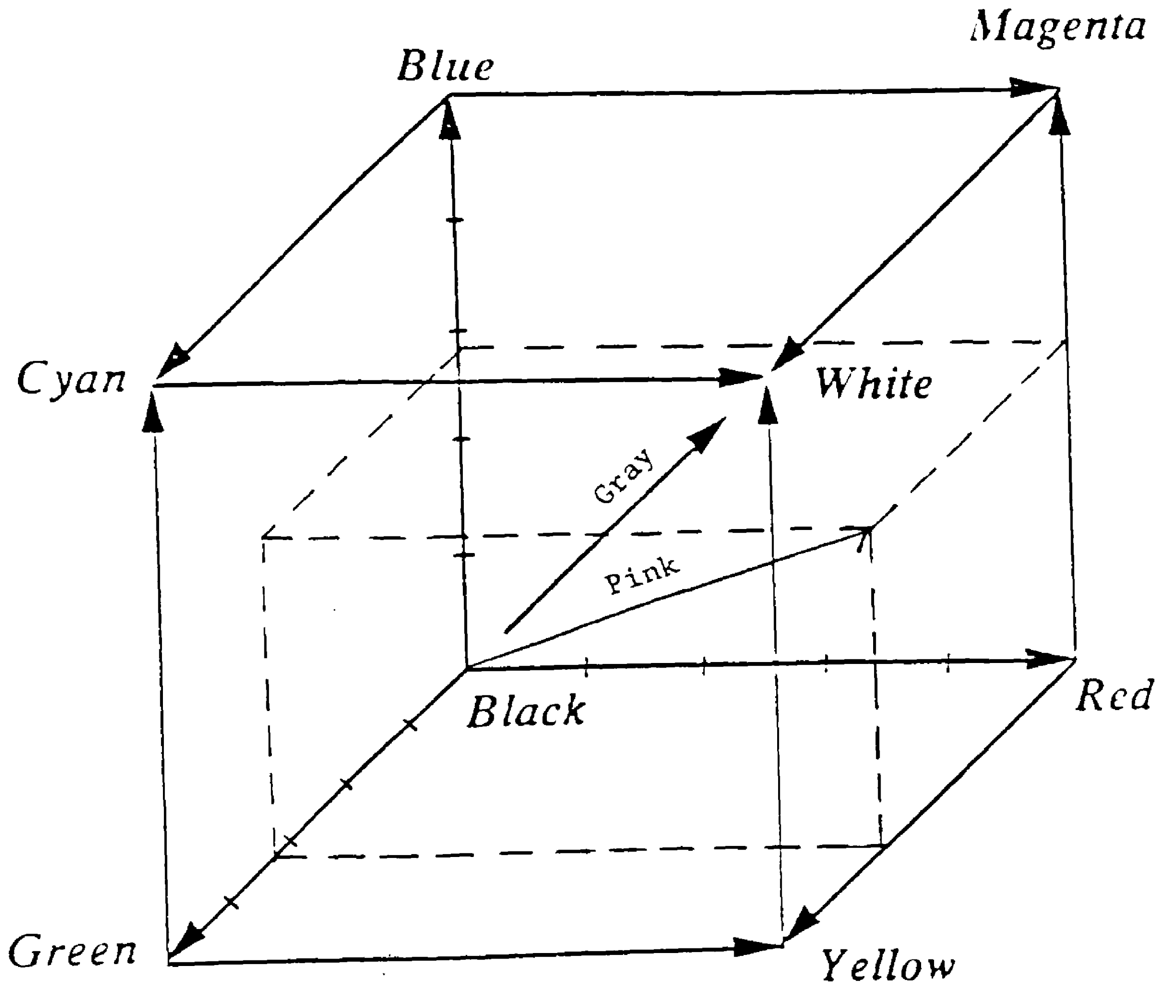


Figure 3-3. Three-dimensional RGB color cube. The origin, black, has values of 0 for all three TM channels used in the color composite, while the values for white are (255, 255, 255). The gray line has an equal proportion of red, green, and blue; the pink does not. Both increase in brightness as distance from the origin increases.


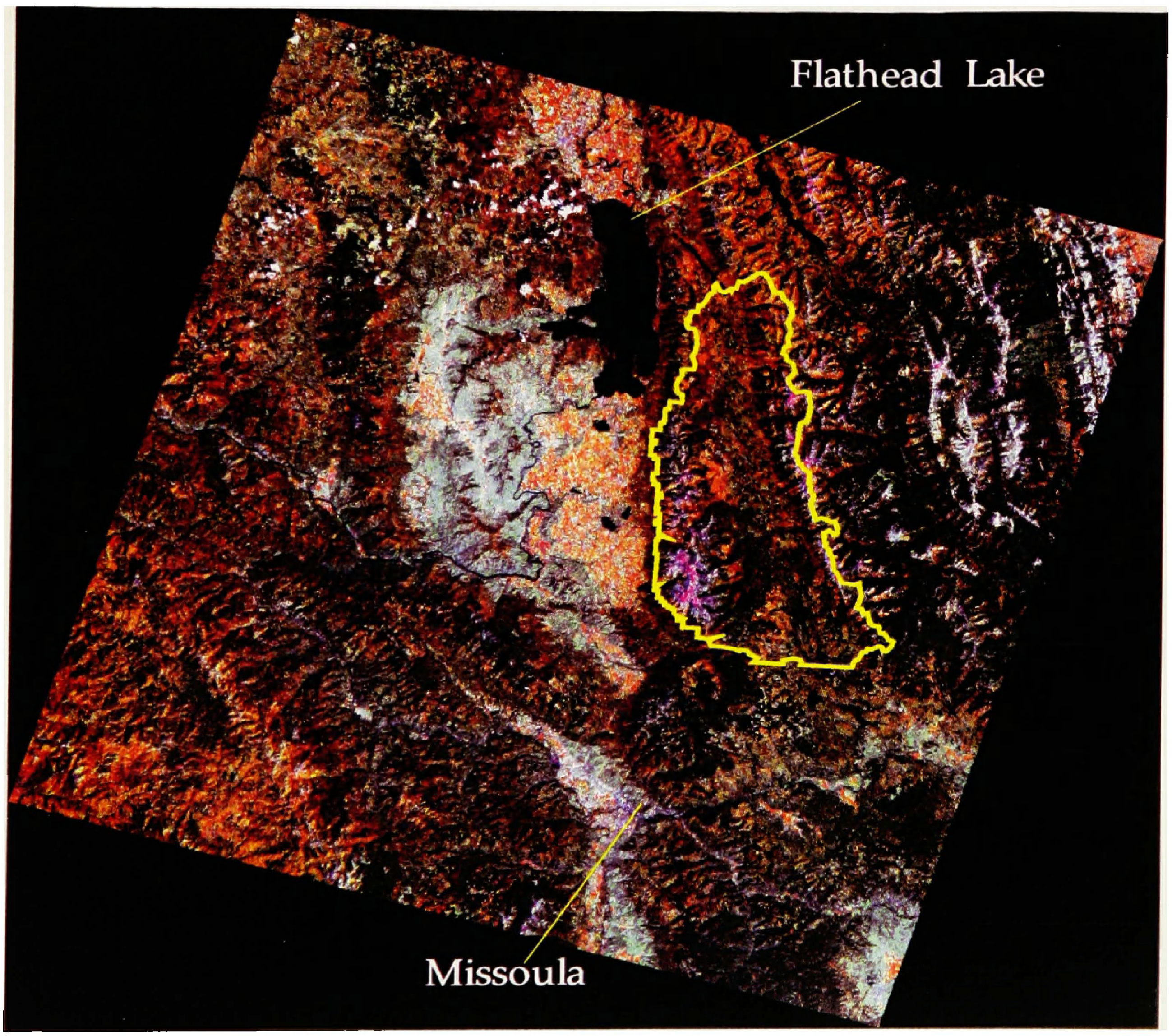


Figure 3-4. Location of the Seeley-Swan landscape within Landsat TM scene P41/R27, acquired 20 July 1991.

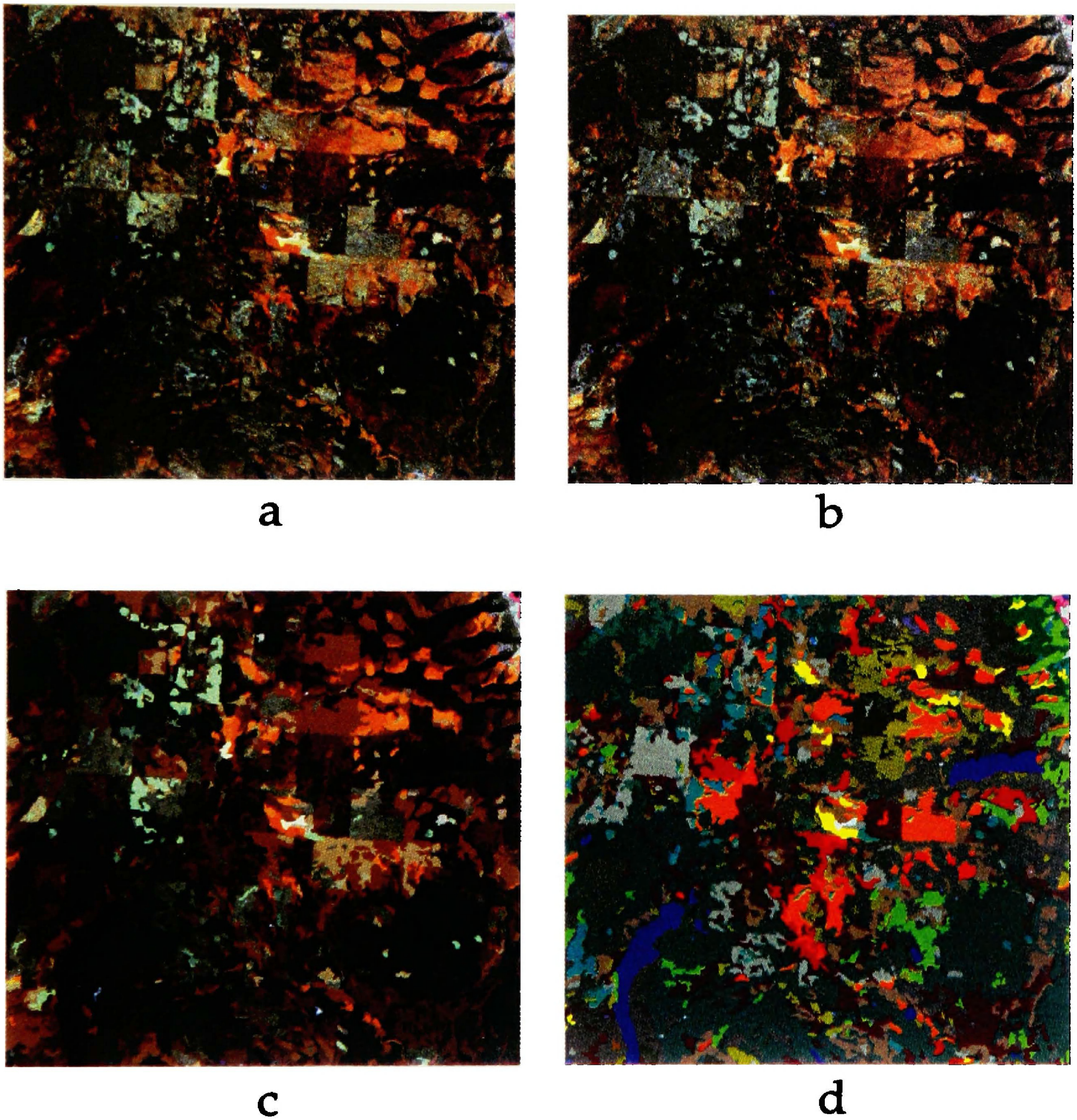


Figure 3-5. Series of 500 pixel by 500 pixel images of southern Swan Valley in northwest Montana; Lindbergh Lake shown in lower left corner. (a) False color composite of TM channels 4, 5, 3 (RGB); (b) unsupervised classification of TM channels 3, 4, 5; (c) unsupervised classification regrouped and merged to 2 ha MMU; (d) classification of land cover types. (From Ma and Redmond, in press.)

Table 3-2. Legend for 1990s vegetation map, Seeley-Swan landscape, northwestern Montana.

## COVER TYPE CODES

For forest types, assigned based on relative percentages of overstory canopy cover (total dominant crown cover of the vegetation polygon).

| | |
|---|---|
| 1 | Water |
| 2 | Snow |
| 3 | Rock |
| 4 | Rocky woodland |
| 5 | Grass |
| 6 | Shrub |
| 7 | Recent cut/seedling plantation |
| 8 | Sapling (Douglas-fir/lodgepole pine/western larch/ponderosa pine) |
| 9 | Sapling (mixed conifer/hardwood) |
| 10 | Douglas-fir |
| 11 | Lodgepole pine |
| 12 | Ponderosa pine |
| 13 | Western larch |
| 14 | Grand fir |
| 15 | Western red cedar |
| 16 | Douglas-fir/western larch |
| 17 | Engelmann spruce/subalpine fir |
| 18 | Whitebark pine/Engelmann spruce/subalpine fir |
| 19 | Mixed conifer |
| 20 | Broadleaf |
| 21 | Sagebrush |
| 22 | Mixed conifer/broadleaf |
| 23 | Wet meadow |
| 24 | Cloud shadow |
| 25 | Cloud |
| 26 | Snowmelt |
| 27 | Barren |
| 28 | Urban |
| 29 | Agriculture |
| 30 | Grass/shrub |

Table 3-2 continued. Legend for 1990s vegetation map, Seeley-Swan landscape, northwestern Montana.

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**CANOPY CLOSURE**

Classified based on CANOPY_COVER field in Swan Lake Ranger District's timber stand database.

| | |
|---|---|
| 1 | Low (0-29%) |
| 2 | Medium (30-59%) |
| 3 | High (60-100%) |

---

**SIZE CLASS**

Classified based on regroupings of STAND_SIZE_CLASS field in Swan Lake Ranger District's timber stand database.

| | |
|---|---|
| 1 | Seedling (NONS and SEED, <1.0" DBH) |
| 2 | Sapling (SAPL, 1.0-4.9" DBH) |
| 3 | Pole (IPOL, POLE, MHRP, and MLRP, 5.0-8.9" DBH) |
| 4 | Immature (IMSA and MULT, 9.0-13.9" DBH) |
| 5 | Mature/overmature (MHRS, MLRS, and SAWT, ≥14.0" DBH) |

---

Table 3-3. Standard codes for cover type (including size class[a]), and the corresponding 1930s and 1990s cover types.

| CODE | | COVER TYPES[b] | | |
|---|---|---|---|---|
| | | Standard | 1930s | 1990s |
| 1. | WAT | water | water | water |
| 2. | BAR | barren | barren | snow, snowmelt, rock, barren |
| 3. | ROC | rocky woodland | all rocky noncommercial and woodland types | rocky woodland |
| 4. | GRA | grass | grass | grass, wet meadow |
| 5. | SHR | shrub | brush | shrub, grass/shrub |
| 6. | B1 | broadleaf pole | cottonwood | broadleaf |
| 7. | B2 | broadleaf m/om | | |
| 8. | URB | urban | townsites | urban |
| 9. | AGR | agriculture | cultivated, stump pasture | agriculture |
| 10. | BU | recent burn | all burn types | recent burn |
| 11. | CUT | recent cut/ seedling/ sapling | all nonstocked cutover types, seedling/ sapling stand class | recent cut/ seedling, sapling types |
| 12. | MC1 | mixed conifer pole | WL/DF, WWP | mixcon, mixcon/broad, GF, DF/WL, WL |
| 13. | MC2 | mixed conifer m/om | | |
| 14. | DF1 | Douglas-fir pole | DF | DF |
| 15. | DF2 | Douglas-fir m/om | | |
| 16. | PP1 | ponderosa pine pole | PP, DF/PP | PP |
| 17. | PP2 | ponderosa pine m/om | | |
| 18. | LP1 | lodgepole pine pole | LP | LP |
| 19. | LP2 | lodgepole pine m/om | | |
| 20. | RC1 | western red cedar pole | WRC | WRC |
| 21. | RC2 | western red cedar m/om | | |
| 22. | SF1 | Engelmann spruce/subalpine fir pole | ES | ES/SAF |
| 23. | SF2 | Engelmann spruce/subalpine fir m/om | | |
| 24. | WBP | whitebark pine/spruce/subalpine fir | subalpine | WBP/ES/SAF |
| 25. | CL | cloud/ cloud shadow | (no comparable type) | cloud, cloud shadow |

[a] pole: 1930s pole, 1990s pole and immature; mature/overmature: 1930s sawtimber, 1990s mature/overmature size classes.

[b] m/om = mature/overmature; DF = Douglas-fir, PP = ponderosa pine, LP = lodgepole pine, WL = western larch, WWP = western white pine, GF = grand fir, WRC = western red cedar, ES = Engelmann spruce, SAF = subalpine fir, WBP = whitebark pine, mixcon = mixed conifer, broad = broadleaf.

Table 3-4. Area occupied by 34 cover types mapped in the 1930s for the Seeley-Swan landscape, northwestern Montana. Total landscape area is approximately 247,925 ha.

| Cover Type | # Polygons | % Area |
|---|---|---|
| Water | 207 | 1.383 |
| Barren | 64 | 7.056 |
| Rocky noncommercial: Douglas-fir | 40 | 0.979 |
| Rocky noncommercial: Douglas-fir/ponderosa pine | 1 | 0.012 |
| Rocky noncommercial: Engelmann spruce | 8 | 0.239 |
| Rocky noncommercial: lodgepole pine | 6 | 0.318 |
| Rocky noncommercial: western larch/Douglas-fir | 15 | 0.602 |
| Woodland: Douglas-fir/ponderosa pine | 1 | 0.158 |
| Woodland: ponderosa pine | 1 | 0.034 |
| Grass | 153 | 0.819 |
| Brush | 77 | 1.326 |
| Cottonwood | 23 | 0.301 |
| Cultivated | 34 | 0.368 |
| Stump pasture | 13 | 0.046 |
| Subalpine: commercial nonstocked burn, pre-1925 | 5 | 0.374 |
| Subalpine: commercial nonstocked burn, post-1925 | 4 | 0.177 |
| Commercial nonstocked burn, pre-1925: Engelmann spruce | 4 | 0.105 |
| Commercial nonstocked burn, pre-1925: lodgepole pine | 2 | 0.019 |
| Commercial nonstocked burn, pre-1925: western larch/Douglas-fir | 8 | 0.506 |
| Commercial nonstocked burn, post-1925: Douglas-fir/ponderosa pine | 2 | 0.032 |
| Commercial nonstocked burn, post-1925: Engelmann spruce | 1 | 0.121 |
| Commercial nonstocked burn, post-1925: western larch/Douglas-fir | 6 | 0.443 |
| Rocky noncommercial: nonstocked burn, pre-1925 | 1 | 0.014 |
| Nonstocked cutover: Douglas-fir/ponderosa pine | 3 | 0.051 |
| Nonstocked cutover: western larch/Douglas-fir | 1 | 0.005 |
| Western larch/Douglas-fir | 325 | 26.806 |
| Western white pine | 29 | 2.244 |
| Douglas-fir | 73 | 2.783 |
| Douglas-fir/ponderosa pine | 114 | 5.359 |
| Ponderosa pine | 19 | 0.731 |
| Lodgepole pine | 247 | 17.269 |
| Western red cedar | 5 | 0.153 |
| Engelmann spruce | 173 | 11.973 |
| Subalpine | 49 | 17.817 |

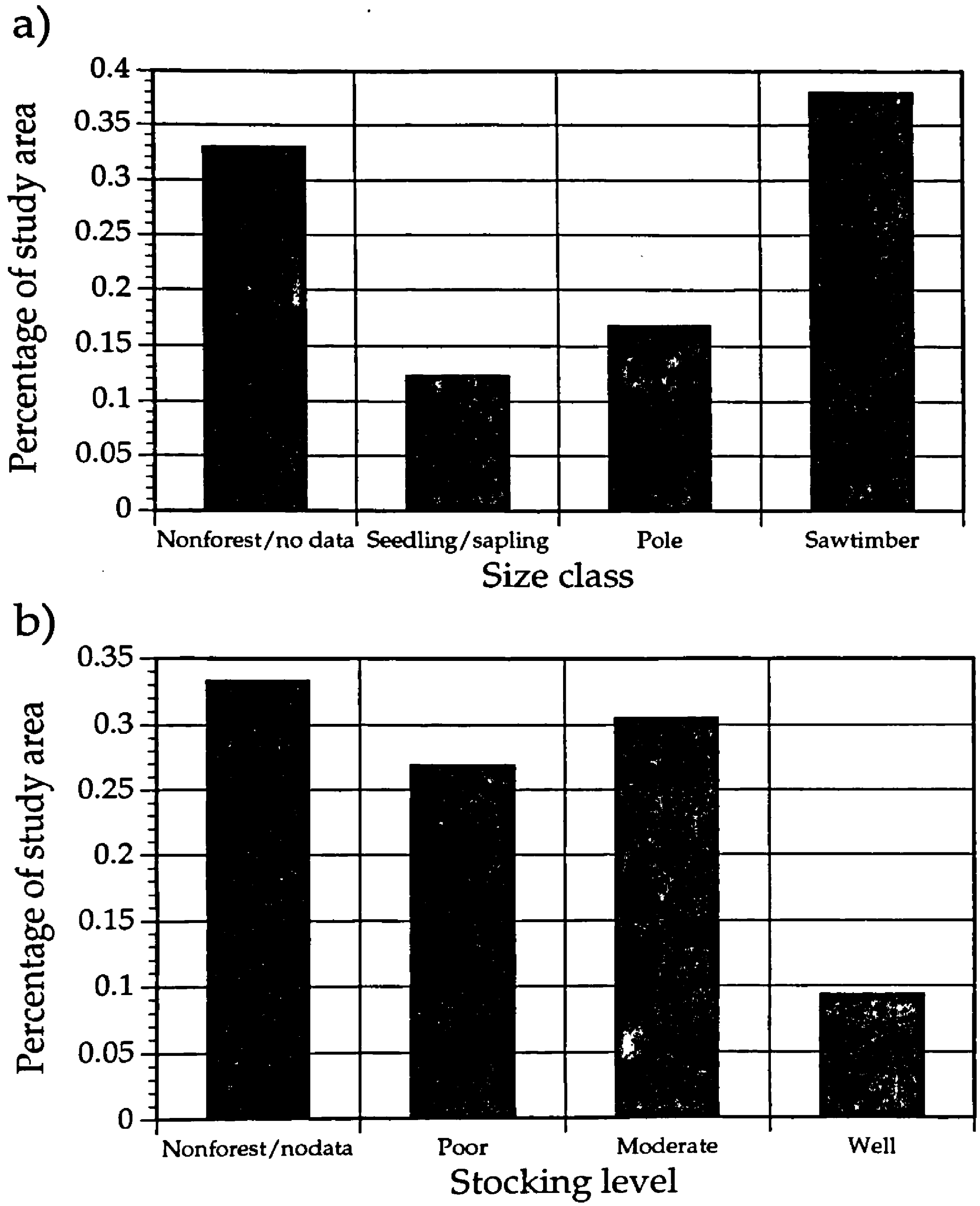


Figure 3-6. Frequency distributions for a) size classes and b) stocking levels as mapped for the 1930s, Seeley-Swan landscape, northwestern Montana.

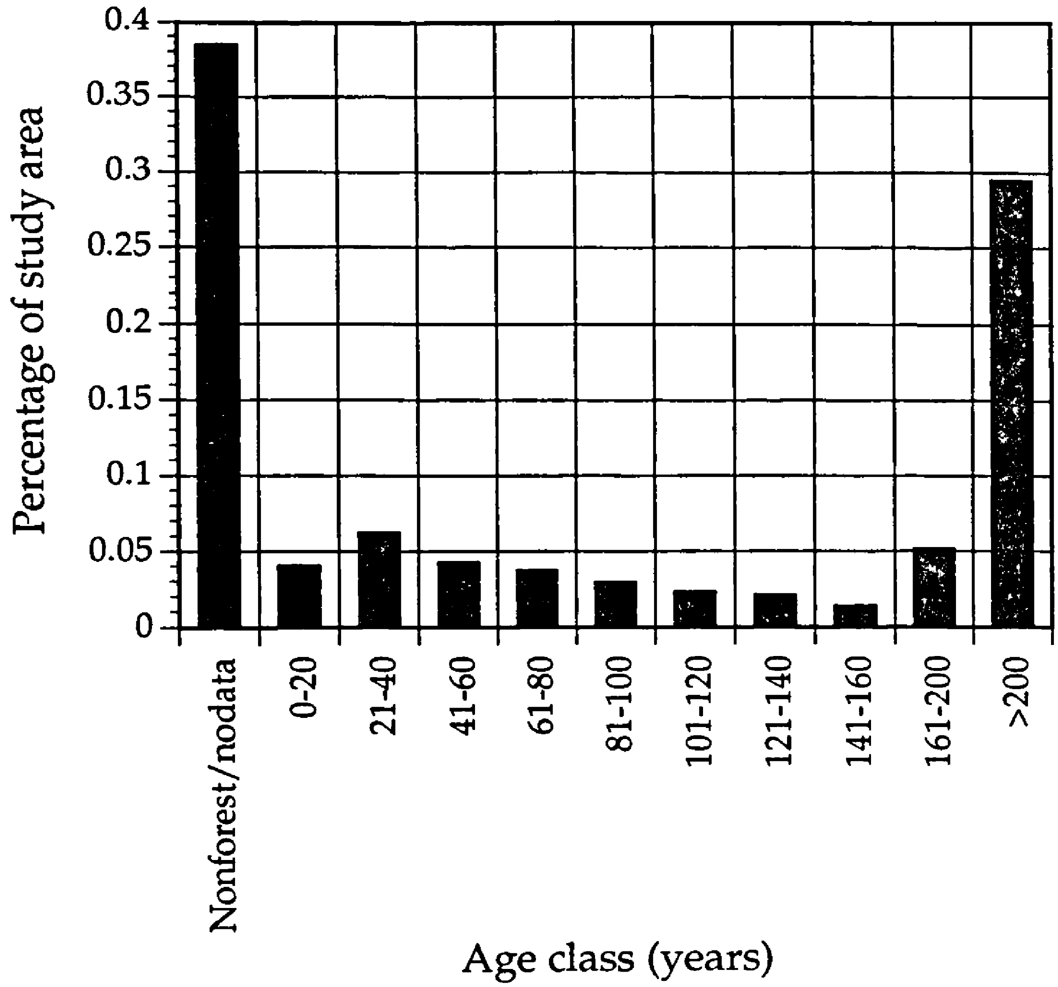


Figure 3-7. Frequency distribution for age classes as mapped for the 1930s, Seeley-Swan landscape, northwestern Montana.

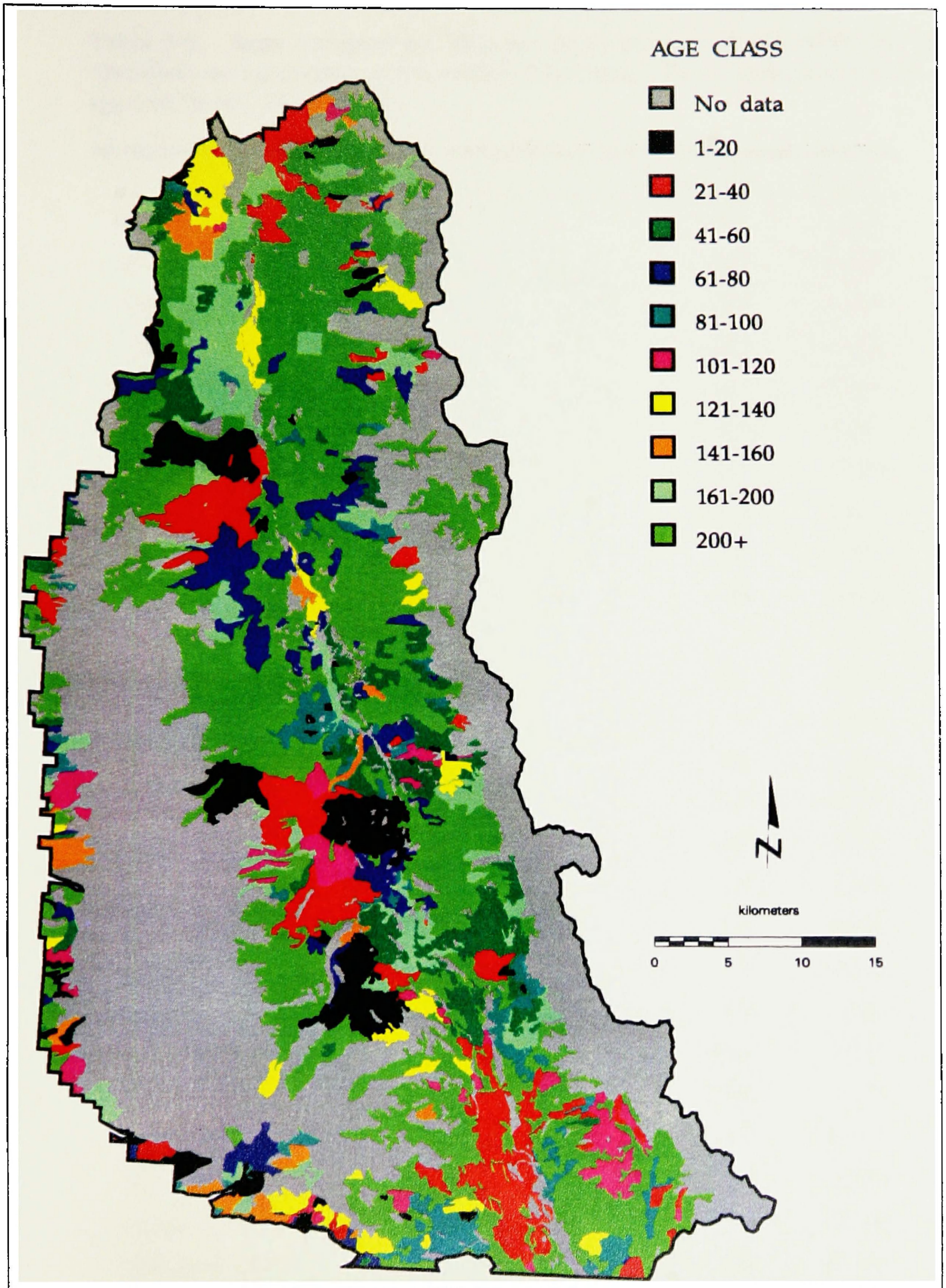


Figure 3-8. Age classes in the Seeley-Swan landscape, northwestern Montana, in the 1930s. Data were typically not recorded for higher-elevation stands; nonforest types are also listed as "no data."

Table 3-5. Area occupied by 30 cover types mapped in the 1990s for the Seeley-Swan landscape, northwestern Montana. Total landscape area is approximately 247,925 ha.

| Cover Type | # Polygons | % Area |
|---|---|---|
| Water | 769 | 1.453 |
| Snow | 307 | 1.789 |
| Snowmelt | 747 | 1.977 |
| Rock | 418 | 0.983 |
| Rocky woodland | 497 | 1.218 |
| Grass | 316 | 0.886 |
| Wet meadow | 539 | 1.520 |
| Shrub | 910 | 3.970 |
| Grass/shrub | 1124 | 3.029 |
| Broadleaf | 151 | 0.489 |
| Urban | 4 | 0.007 |
| Agricultural | 221 | 0.549 |
| Recent burn | 25 | 0.078 |
| Recent cut/seedling | 842 | 2.824 |
| Sapling | 2355 | 12.351 |
| Mixed conifer | 3260 | 15.348 |
| Mixed conifer/broadleaf | 136 | 0.942 |
| Grand fir | 231 | 0.789 |
| Douglas-fir/western larch | 498 | 2.514 |
| Western larch | 696 | 2.540 |
| Douglas-fir | 1706 | 6.544 |
| Ponderosa pine | 446 | 1.737 |
| Lodgepole pine | 1713 | 10.805 |
| Western red cedar | 659 | 2.285 |
| Engelmann spruce/subalpine fir | 2616 | 12.500 |
| Whitebark pine/Engelmann spruce/subalpine fir | 3354 | 10.481 |
| Cloud, cloud shadow | 363 | 0.390 |

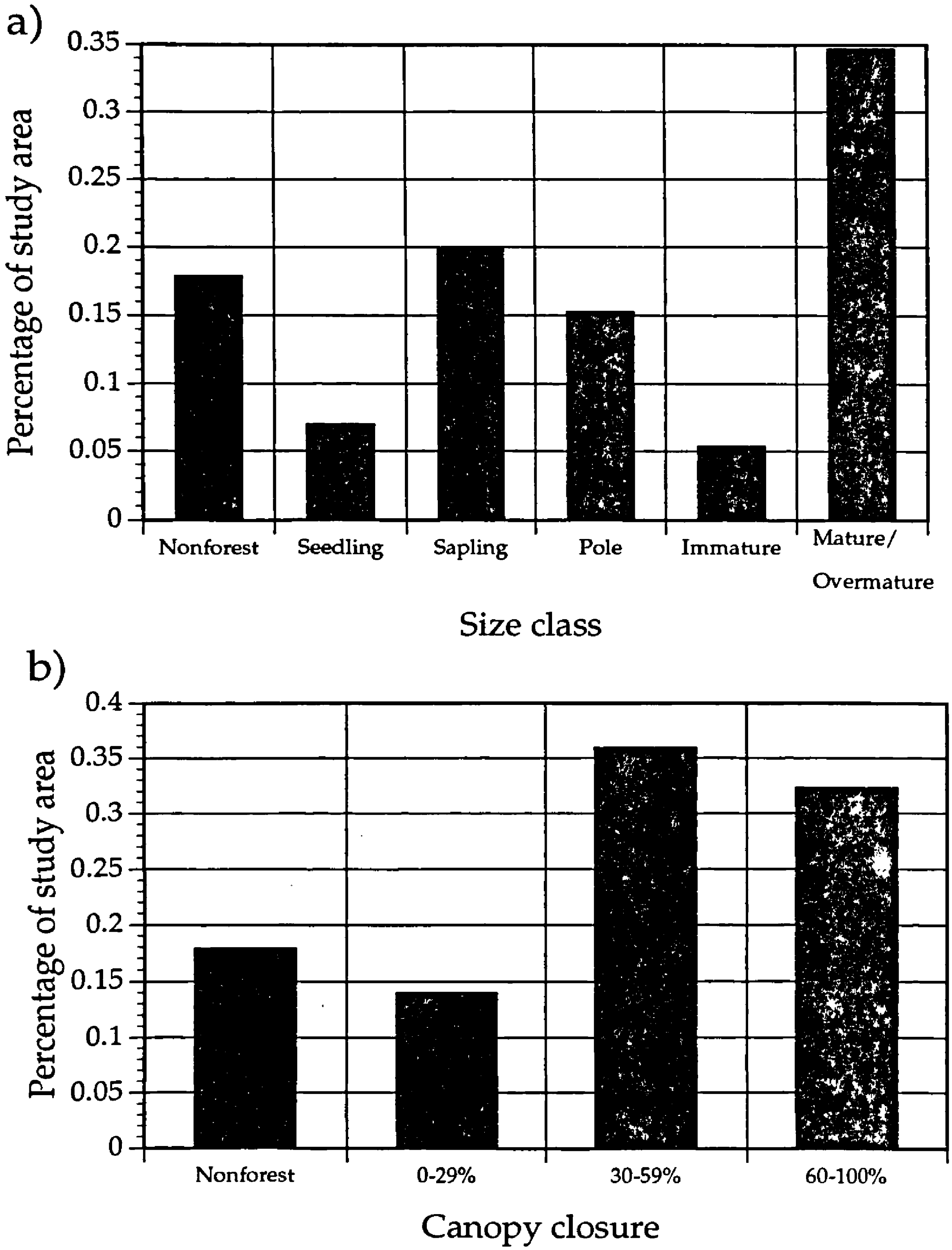


Figure 3-9. Frequency distributions for a) size classes and b) levels of canopy closure as mapped for the 1990s, Seeley-Swan landscape, northwestern Montana.

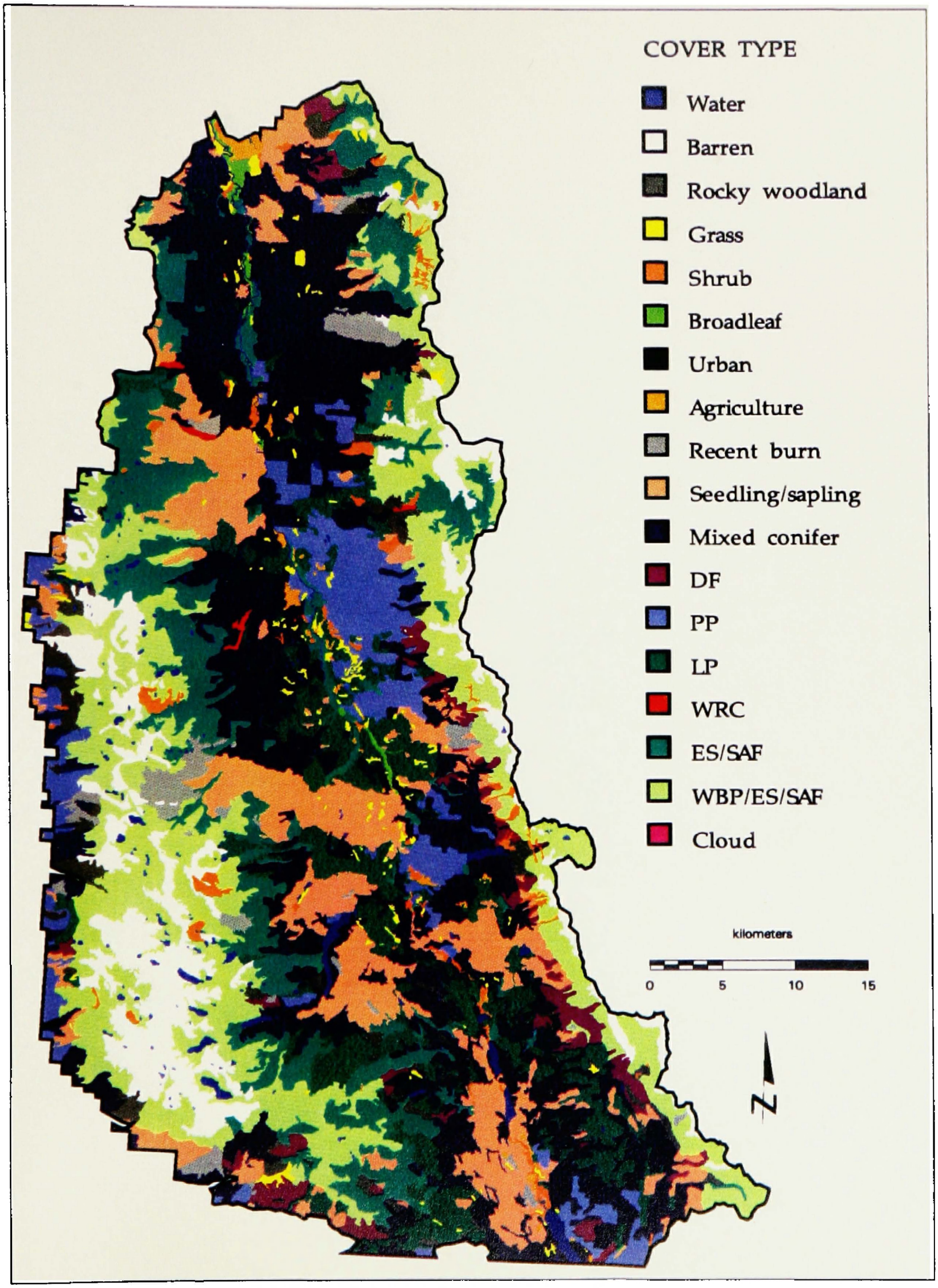


Figure 3-10. Standardized cover types in the Seeley-Swan landscape, northwestern Montana, in the 1930s (16 ha MMU).

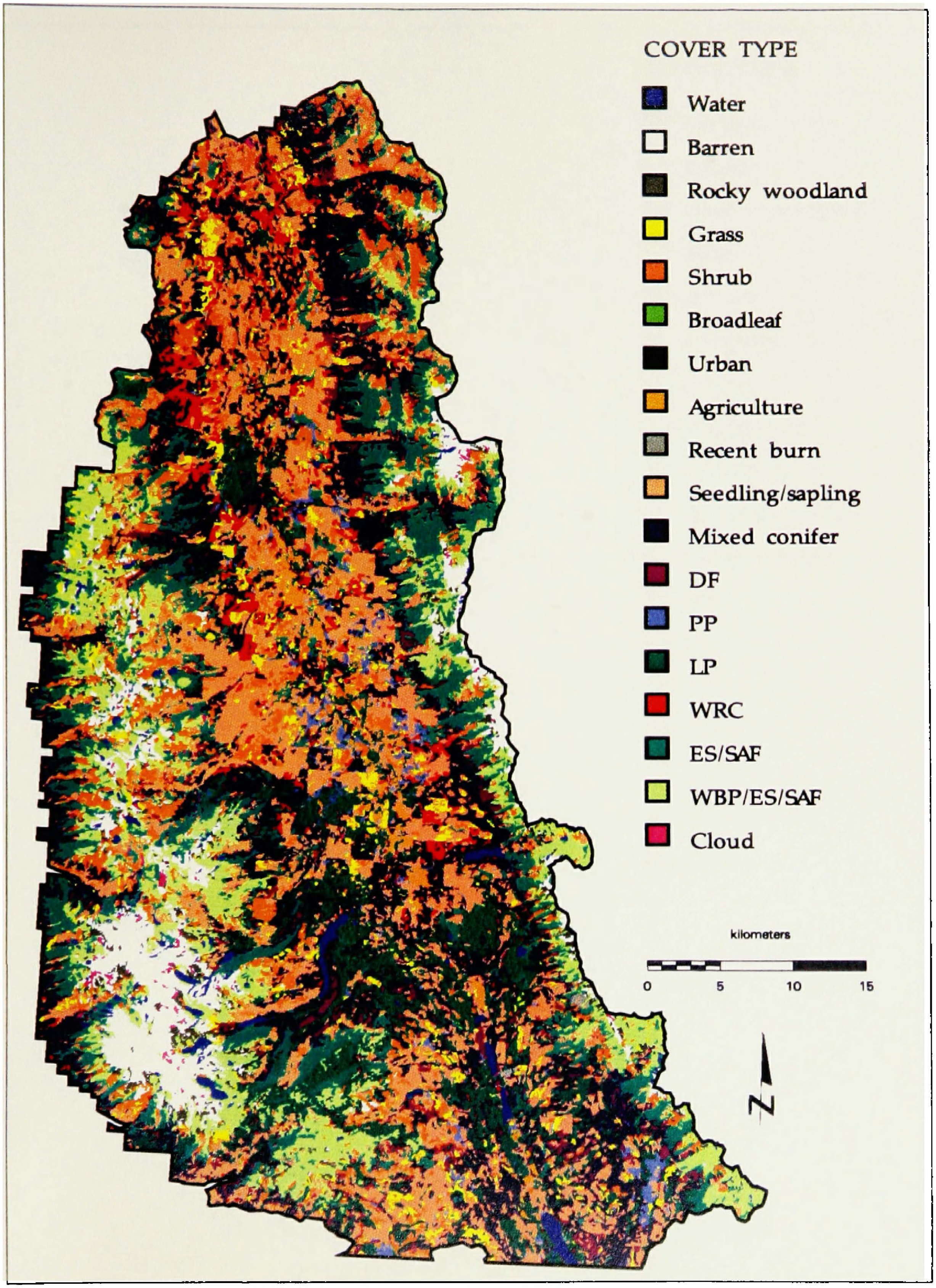


Figure 3-11. Standardized cover types in the Seeley-Swan landscape, northwestern Montana, in the 1990s (2 ha MMU).

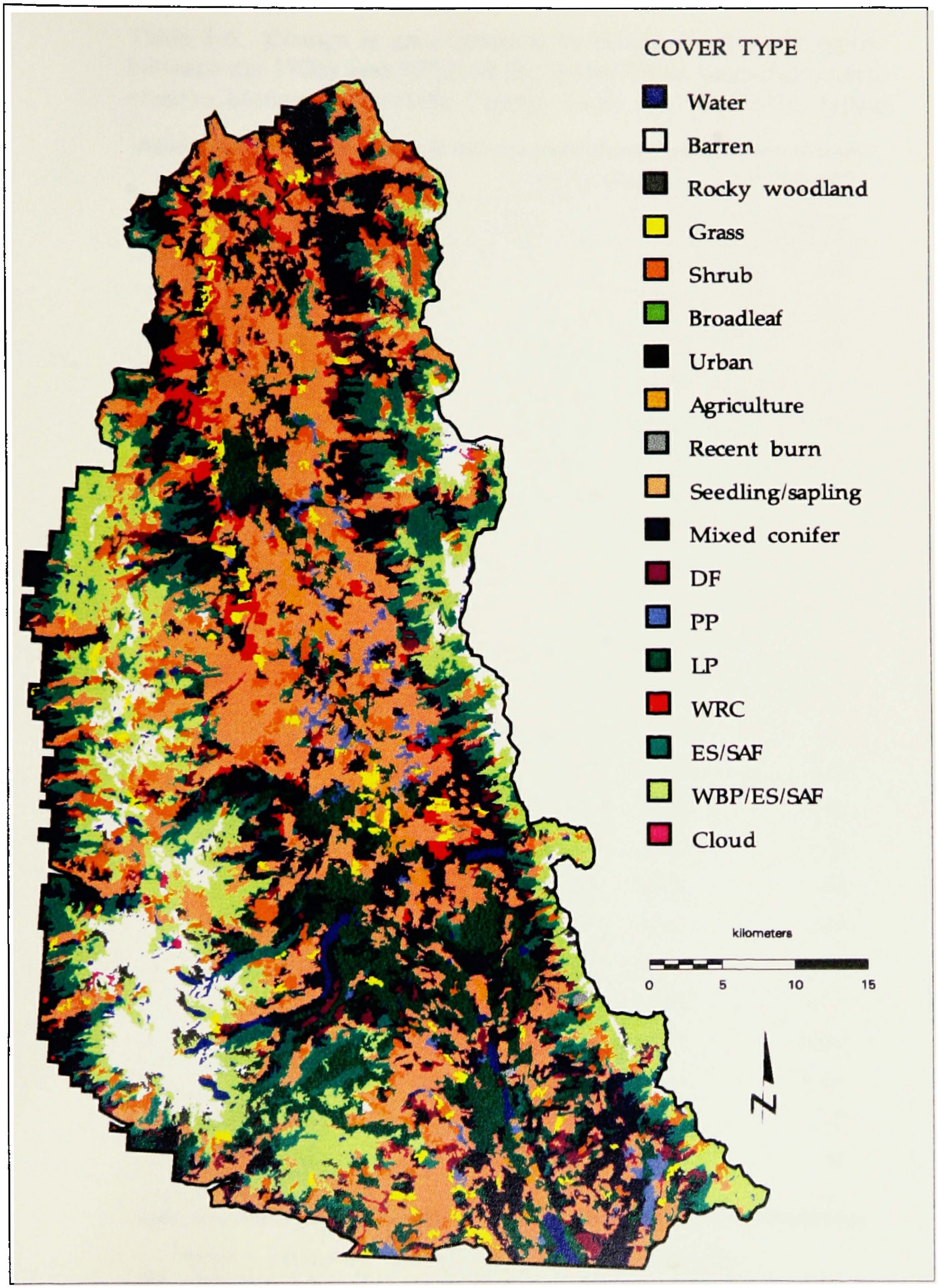


Figure 3-12. Standardized cover types in the Seeley-Swan landscape, northwestern Montana, in the 1990s (16 ha MMU).

Table 3-6. Change in area occupied by standardized cover types between the 1930s and 1990s in the Seeley-Swan landscape, northwestern Montana (2 ha MMU water, 16 ha MMU all other types).

| | HECTARES | | PERCENT |
|---|---|---|---|
| COVER TYPE[a] | 1930s | 1990s | CHANGE |
| 1. Water | 3430 | 3603 | +5 |
| 2. Barren | 17492 | 11857 | -32 |
| 3. Rocky woodland | 5807 | 2040 | -65 |
| 4. Grass | 2031 | 3477 | +71 |
| 5. Shrub | 3287 | 13892 | +323 |
| 6. Broadleaf pole | 746 | 0 | -100 |
| 7. Broadleaf m/om | 0 | 0 | 0 |
| 8. Urban | 0 | 17 | ++ |
| 9. Agriculture | 1027 | 1219 | +19 |
| 10. Burn | 4439 | 149 | -97 |
| 11. Seedling/sapling | 30405 | 64489 | +112 |
| 12. Mixed conifer pole | 8823 | 20725 | +135 |
| 13. Mixed conifer m/om | 53125 | 27291 | -49 |
| 14. DF pole | 3840 | 2001 | -48 |
| 15. DF m/om | 2103 | 5165 | +146 |
| 16. PP pole | 1049 | 1355 | +29 |
| 17. PP m/om | 13995 | 1686 | -88 |
| 18. LP pole | 23318 | 11935 | -49 |
| 19. LP m/om | 63 | 16840 | +26516 |
| 20. WRC pole | 0 | 1050 | ++ |
| 21. WRC m/om | 311 | 3645 | +1070 |
| 22. ES/SAF pole | 3846 | 6512 | +69 |
| 23. ES/SAF m/om | 24601 | 19086 | -22 |
| 24. WBP/ES/SAF | 44179 | 29668 | -33 |
| 25. Cloud | 0 | 229 | ++ |

[a]m/om = mature/overmature size class; DF = Douglas-fir, PP = ponderosa pine, LP = lodgepole pine, WRC = western redcedar, ES = Engelmann spruce, SAF = subalpine fir, WBP = whitebark pine.

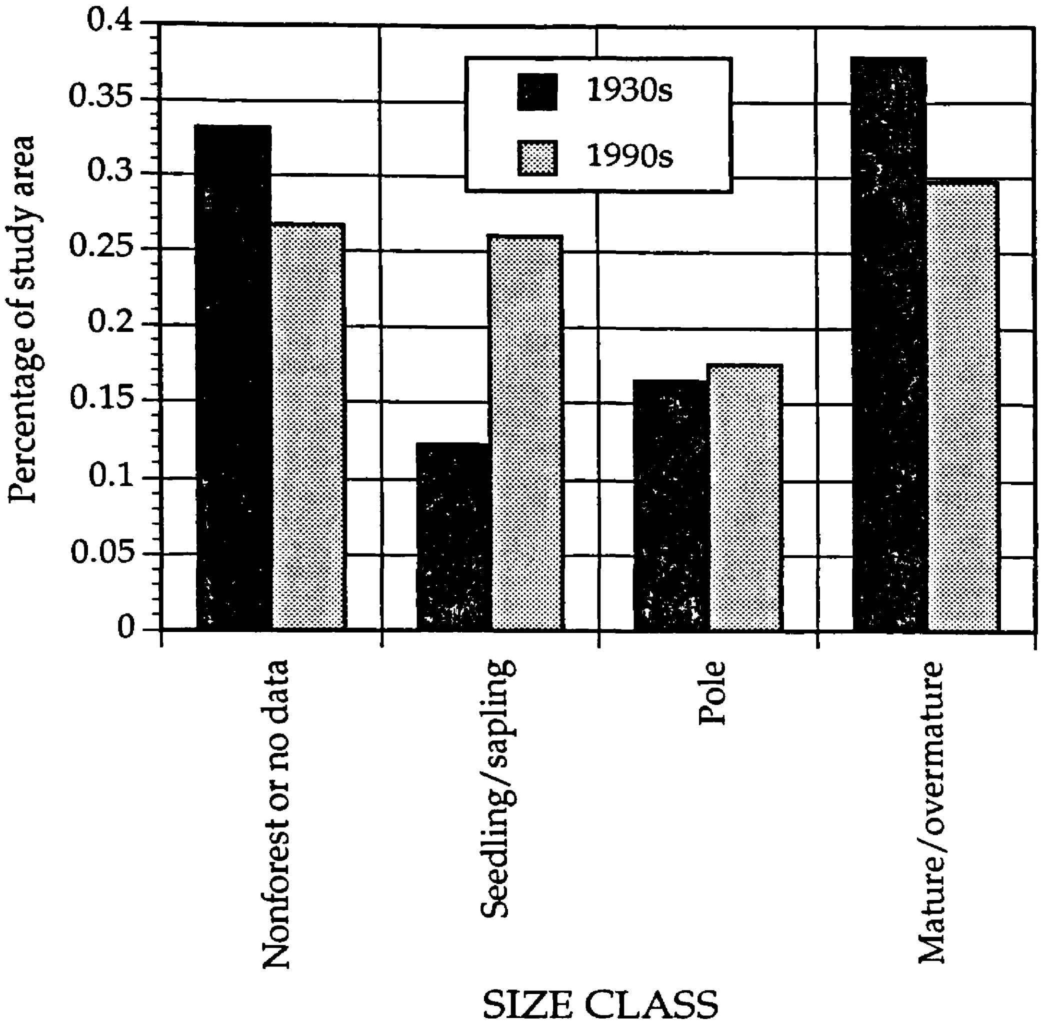


Figure 3-13. Comparison of size class distributions, 1930s versus 1990s, Seeley-Swan landscape, northwestern Montana (16 ha MMU).

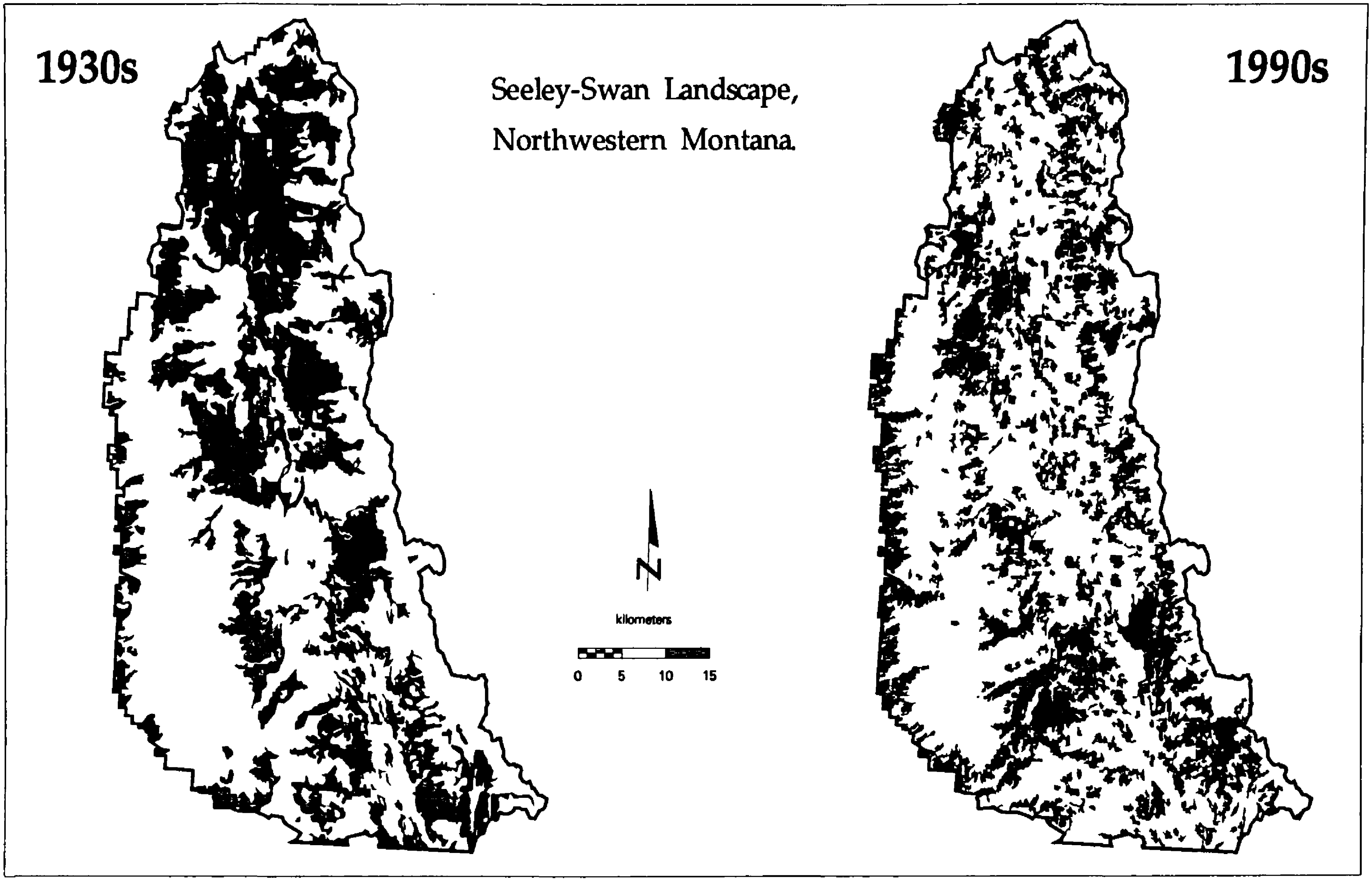


Figure 3-14. Spatial distribution of the mature/overmature forest size class, 1930s and 1990s (16 ha MMU).

Table 3-7. Landscape metrics by cover type for the 1930s and 1990s, Seeley-Swan landscape[a], northwestern Montana.

| | PATCHES[b] | | SIZE | | SHAPE | | CORE | | NEIGHBOR | |
|---|---|---|---|---|---|---|---|---|---|---|
| TYPE | 1930s | 1990s | 1930s | 1990s | 1930s | 1990s | 1930s | 1990s | 1930s | 1990s |
| WAT | 197 | 761 | 17.4 (52.1) | 4.7 (22.2) | 1.49 | 1.28 | 22.13 | 6.35 | 779 (1012) | 385 (681) |
| BAR | 71 | 65 | 246.4 (984.3) | 182.4 (779.4) | 2.07 | 2.57 | 42.67 | 43.20 | 389 (548) | 610 (1441) |
| ROC | 65 | 57 | 89.3 (122.0) | 35.8 (22.4) | 1.95 | 2.44 | 53.11 | 39.08 | 1192 (1479) | 991 (1604) |
| GRA | 153 | 89 | 13.3 (17.5) | 39.1 (28.1) | 1.58 | 2.27 | 24.70 | 42.98 | 916 (1808) | 1561 (1338) |
| SHR | 78 | 279 | 42.1 (54.4) | 49.8 (47.1) | 2.04 | 2.35 | 37.39 | 44.77 | 1524 (1840) | 639 (743) |
| B1 | 22 | 0 | 33.9 (69.5) | 0 | 2.03 | 0 | 19.90 | 0 | 789 (1696) | 0 |
| URB | 0 | 1 | 0 | 17.37 (0) | 0 | 2.63 | 0 | 21.76 | 0 | -- |
| AGR | 43 | 31 | 23.9 (50.7) | 39.3 (29.6) | 1.56 | 2.10 | 32.29 | 44.66 | 1082 (1352) | 1305 (1468) |
| BU | 28 | 3 | 158.5 (286.5) | 49.8 (26.6) | 1.71 | 2.04 | 57.45 | 53.57 | 3947 (3906) | 18409 (17961) |
| CUT | 104 | 413 | 292.4 (829.6) | 156.2 (637.1) | 1.97 | 2.79 | 55.56 | 47.79 | 678 (885) | 197 (284) |
| MC1 | 69 | 270 | 127.9 (240.5) | 76.8 (122.0) | 1.82 | 2.76 | 58.12 | 42.60 | 1079 (1045) | 503 (528) |
| MC2 | 101 | 295 | 526.0 (1666.8) | 92.5 (163.2) | 2.33 | 2.79 | 57.48 | 44.25 | 541 (1131) | 369 (449) |
| DF1 | 29 | 57 | 132.4 (177.9) | 35.1 (28.2) | 1.92 | 2.52 | 64.24 | 34.81 | 2788 (5844) | 2543 (2154) |
| DF2 | 19 | 128 | 110.7 (132.4) | 40.4 (28.2) | 1.77 | 2.49 | 62.28 | 40.24 | 2451 (2715) | 1297 (1302) |
| PP1 | 26 | 26 | 40.4 (57.1) | 52.1 (70.6) | 1.62 | 2.64 | 31.92 | 37.75 | 2435 (8094) | 2388 (1926) |
| PP2 | 53 | 29 | 264.1 (703.1) | 58.1 (45.2) | 1.82 | 2.72 | 61.43 | 44.42 | 1085 (2041) | 1518 (2241) |
| LP1 | 97 | 126 | 240.4 (352.6) | 94.7 (244.8) | 2.10 | 2.70 | 64.74 | 41.85 | 801 (1576) | 1144 (1512) |
| LP2 | 1 | 125 | 63.3 (0) | 134.7 (361.1) | 1.87 | 2.97 | 61.17 | 43.69 | -- | 752 (1237) |
| RC1 | 0 | 30 | 0 | 35.0 (21.1) | 0 | 2.31 | 0 | 41.39 | 0 | 1514 (1410) |
| RC2 | 4 | 60 | 77.9 (28.3) | 60.8 (75.6) | 2.1 | 2.49 | 58.04 | 43.46 | 8130 (3960) | 1224 (1218) |
| SF1 | 42 | 164 | 91.6 (96.8) | 39.7 (29.5) | 1.84 | 2.50 | 54.03 | 38.43 | 1569 (2944) | 948 (1082) |
| SF2 | 68 | 214 | 361.8 (622.3) | 89.2 (109.1) | 2.35 | 2.67 | 59.18 | 46.08 | 1059 (2125) | 533 (780) |
| WBP | 50 | 140 | 883.6 (1725.5) | 211.9 (442.6) | 2.81 | 3.18 | 56.06 | 50.13 | 531 (1471) | 174 (294) |
| CL | 0 | 7 | 0 | 32.7 (9.7) | 0 | 2.47 | 0 | 38.63 | 0 | 10039 (10668) |

[a] 16 ha MMU for all cover types except water (2 ha MMU); cover type B2 absent from the landscapes at this MMU.
[b] Patches = number of patches, size = mean patch size (standard deviation), shape = mean shape index, core = mean core area index, nearest neighbor = mean nearest neighbor distance (standard deviation).

Table 3-8. Crown competition factor (CCF, Krajicek et al. 1961) calculations for 5 stand types mapped in the 1930s. Dominant species, age class, stocking level, and site class were attributes mapped in the 1930s; normal stand volume, trees per acre, and DBH average tree were taken from Haig (1932) and Meyer (1938). Number of trees per diameter class (not shown) was also estimated from tables in Haig (1932) and Meyer (1938), then used in CCF calculations based on Wyckoff et al. (1982). CCF values were adjusted to compensate for lower-than-normal stocking levels (CCF * normal volume / estimated volume) using both the highest point and the midpoint in the estimated stocking level.

| DOMINANT SPECIES | AGE CLASS (years) | STOCKING LEVEL (mbf) | SITE CLASS | NORMAL STAND VOLUME (mbf) | TREES PER ACRE | DBH AVE. TREE (in) | TOTAL CCF (NORMAL STAND) | ADJUSTED CCF (HIGH) | ADJUSTED CCF (MID) |
|---|---|---|---|---|---|---|---|---|---|
| Western white pine | 120-140 | 4-10 | 50 | 37.0 | 630 | 9.7 | 256.3 | 69.2 | 48.7 |
| Western white pine | 140-160 | 10-20 | 50 | 43.8 | 600 | 10.2 | 243.2 | 111.9 | 82.7 |
| Ponderosa pine | 100-120 | 3-7 | 90 | 31.1 | 197 | 14.1 | 194.9 | 44.8 | 31.2 |
| Ponderosa pine | 200+ | 3-7 | 80 | 43.0 | 92 | 19.9 | 167.6 | 26.8 | 20.1 |
| Ponderosa pine | 200+ | 7-13 | 90 | 54.2 | 79 | 22.2 | 179.0 | 43.0 | 32.2 |

Table 3-9. A comparison of descriptive landscape statistics for the Seeley-Swan study area (247,925 ha), northwestern Montana, across two time periods. A minimum mapping unit of 16 ha was used for both periods for all cover types except water (2 ha).

| LANDSCAPE MEASURE | 1930s | 1990s | P Value[a] |
|---|---|---|---|
| Number of patches | 1320 | 3370 | 0.0679 |
| Mean patch size (ha) | 187.82 | 73.57 | 0.2104 |
| Patch size standard deviation (ha) | 728.25 | 291.43 | 0.1894 |
| Largest patch index (%) | 5.30 | 4.41 | 0.3508 |
| Mean shape index | 1.90 | 2.35 | ≤0.001 |
| Mean patch fractal dimension | 1.09 | 1.12 | ≤0.001 |
| Total edge (m) | 5,972,070 | 13,278,390 | 0.1070 |
| Total core area (ha) | 197,425.35 | 149,110.11 | 0.9227 |
| Number of core areas | 2003 | 8327 | 0.0078 |
| Mean core area index (%) | 44.98 | 35.46 | 0.1348 |
| Mean proximity index | 906.19 | 1101.47 | 0.7701 |
| Mean nearest neighbor (m) | 1072.35 | 672.78 | 0.9424 |
| Nearest neighbor standard deviation (m) | 2282.47 | 1435.08 | 0.2695 |
| Patch richness | 21 | 23 | - |
| Shannon's diversity index | 2.37 | 2.45 | - |
| Simpson's diversity index | 0.88 | 0.88 | - |
| Shannon's evenness index | 0.78 | 0.78 | - |
| Simpson's evenness index | 0.92 | 0.92 | - |
| Interspersion/juxtaposition index (%) | 75.74 | 73.78 | 0.1298 |
| Contagion (%) | 57.23 | 53.69 | - |

[a] Derived through Mann-Whitney U tests evaluating for differences in median values for all classes in the 1930s and 1990s landscapes. Some metrics are not calculated (or meaningful) except at the landscape level; thus, P values were not obtained.

Table 3-10. A comparison of descriptive landscape metrics for the Seeley-Swan study area, northwestern Montana, under varying treatments of water patches. Patches of water were either held to a 2 ha MMU and included in the calculations; held to a 2 ha MMU and treated as background (excluded); or allowed to merge to the 100 ha MMU along with all other classes, then included in the calculations. The background method was chosen for further analyses.

| | TREATMENT OF WATER IN ANALYSIS[a] | | |
|---|---|---|---|
| LANDSCAPE MEASURE | Included | Background | Merged |
| Number of patches | 1303 | 542 | 543 |
| Mean patch size (ha) | 190.28 | 450.79 | 456.60 |
| Patch size standard deviation (ha) | 829.03 | 1239.11 | 1224.22 |
| Largest patch index (%) | 8.62 | 8.75 | 8.58 |
| Mean shape index | 2.26 | 3.63 | 3.64 |
| Mean patch fractal dimension | 1.09 | 1.17 | 1.17 |
| Total edge (m) | 8,554,890 | 7,933,020 | 8,234,160 |
| Total core area (ha) | 181,906.38 | 180,176.04 | 183,990.78 |
| Number of core areas | 3462 | 3199 | 3194 |
| Mean core area index (%) | 30.19 | 63.65 | 65.86 |
| Mean proximity index | 23.85 | ∞ | 44.37 |
| Mean nearest neighbor (m) | 976.90 | 1801.10 | 1991.00 |
| Nearest neighbor standard deviation (m) | 2955.12 | 4381.35 | 4904.13 |
| Patch richness | 21 | 20 | 21 |
| Shannon's diversity index | 2.25 | 2.20 | 2.23 |
| Simpson's diversity index | 0.85 | 0.85 | 0.85 |
| Shannon's evenness index | 0.74 | 0.73 | 0.73 |
| Simpson's evenness index | 0.89 | 0.89 | 0.89 |
| Interspersion/juxtaposition index (%) | 70.07 | 68.84 | 68.64 |
| Contagion (%) | 58.07 | 58.41 | 58.44 |

[a] In all treatments, 100 ha MMU for all classes except water.

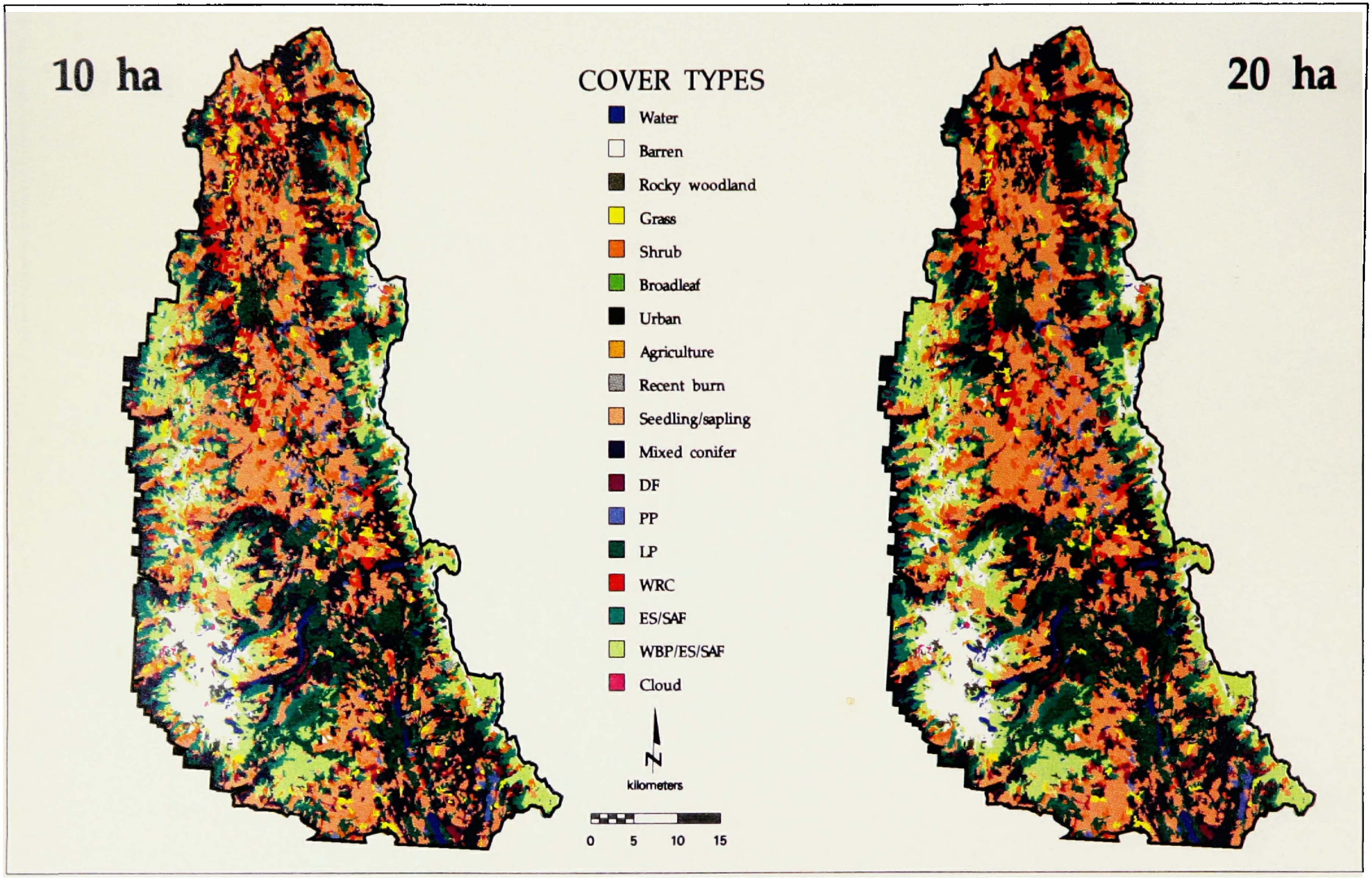


Figure 3-15a. Standardized cover types for 1990s vegetation in the Seeley-Swan landscape, northwestern Montana, as minimum mapping unit increases (10 and 20 ha).

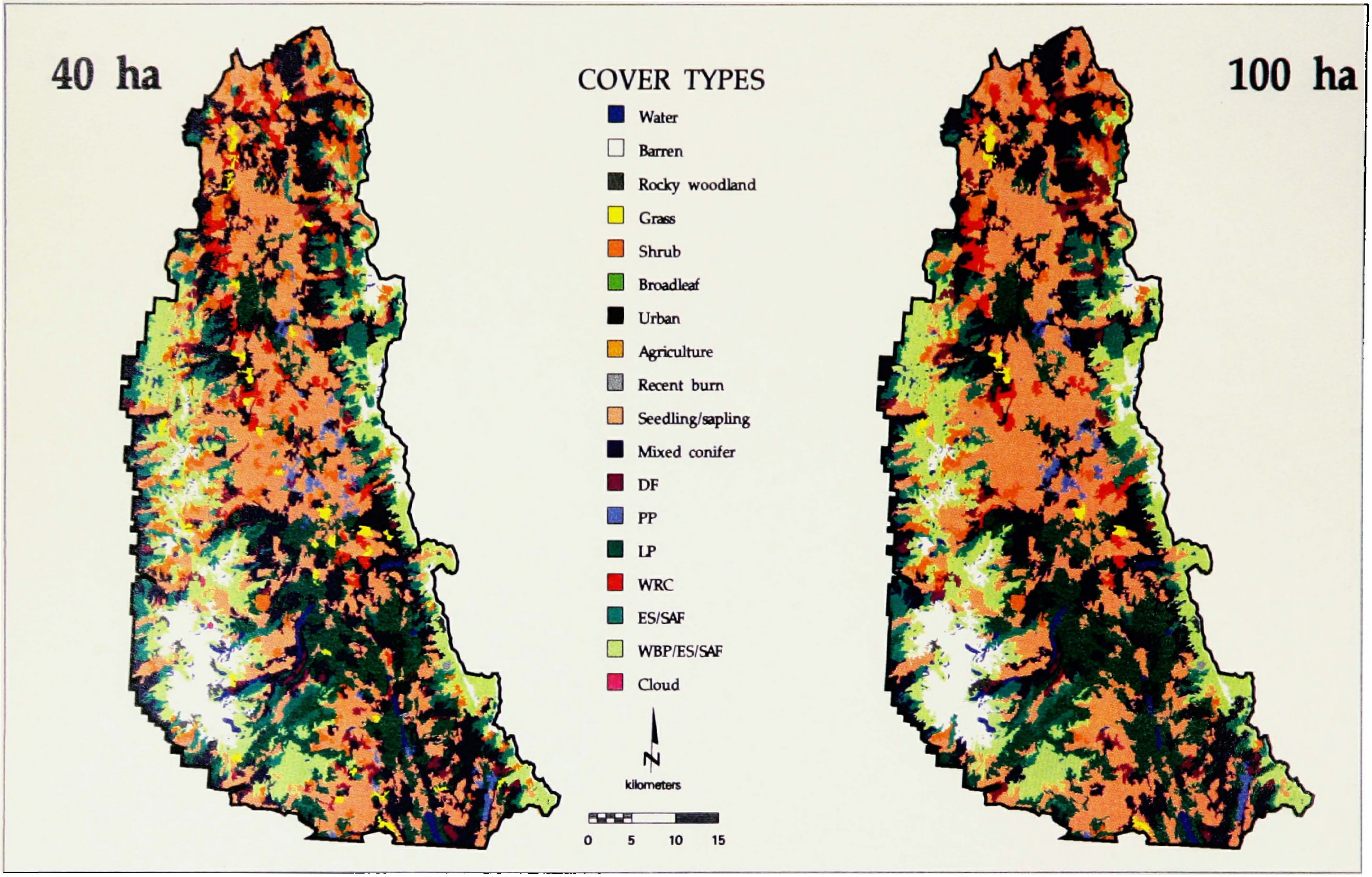


Figure 3-15b. Standardized cover types for 1990s vegetation in the Seeley-Swan landscape, northwestern Montana, as minimum mapping unit increases (40 and 100 ha).

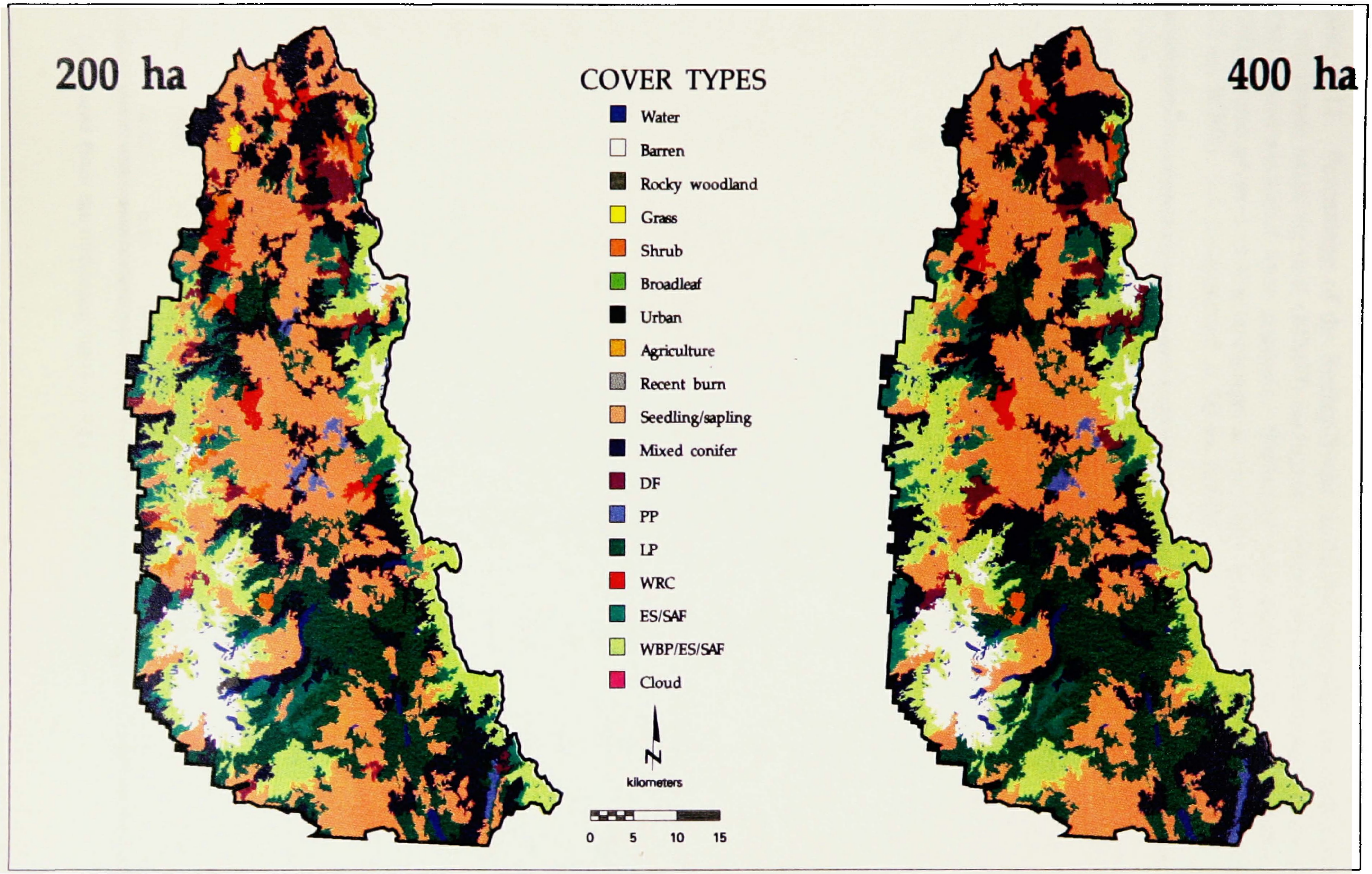


Figure 3-15c. Standardized cover types for 1990s vegetation in the Seeley-Swan landscape, northwestern Montana, as minimum mapping unit increases (200 and 400 ha).

Table 3-11. Percentage of the Seeley-Swan landscape occupied by each cover type as minimum mapping unit (MMU) increases. Water was held constant for all MMUs and excluded from analysis. Broadleaf pole stands (B1) and urban areas (URB) were present in the landscape at very low levels (15 and 17 ha respectively) at 2 ha MMU, but disappeared at 10 ha MMU.

| COVER | MINIMUM MAPPING UNIT | | | | | | | | Percent |
|---|---|---|---|---|---|---|---|---|---|
| TYPE | 2 ha | 10 ha | 16 ha | 20 ha | 40 ha | 100 ha | 200 ha | 400 ha | Change[a] |
| BAR | 4.82 | 4.84 | 4.85 | 4.88 | 4.83 | 4.76 | 4.66 | 4.42 | -8 |
| ROC | 1.24 | 0.90 | 0.84 | 0.80 | 0.58 | 0.20 | 0.11 | 0 | -100 |
| GRA | 2.44 | 1.69 | 1.42 | 1.33 | 1.00 | 0.48 | 0.10 | 0 | -100 |
| SHR | 7.10 | 6.16 | 5.69 | 5.45 | 4.79 | 3.13 | 1.40 | 0.83 | -88 |
| B1 | <.01 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -100 |
| B2 | 0.02 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -100 |
| URB | <.01 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -100 |
| AGR | 0.56 | 0.53 | 0.50 | 0.44 | 0.37 | 0.10 | 0 | 0 | -100 |
| BU | 0.08 | 0.07 | 0.06 | 0.07 | 0.06 | 0.04 | 0 | 0 | -100 |
| CUT | 23.23 | 25.63 | 26.39 | 26.79 | 27.93 | 31.01 | 32.02 | 31.25 | +35 |
| MC1 | 8.29 | 8.40 | 8.48 | 8.57 | 8.82 | 8.59 | 8.05 | 8.17 | -1 |
| MC2 | 10.74 | 10.95 | 11.17 | 11.14 | 11.32 | 10.82 | 11.02 | 11.35 | +6 |
| DF1 | 1.27 | 0.92 | 0.82 | 0.75 | 0.62 | 0.43 | 0.17 | 0 | -100 |
| DF2 | 2.40 | 2.22 | 2.11 | 2.18 | 2.06 | 2.15 | 2.33 | 2.57 | +7 |
| PP1 | 0.70 | 0.55 | 0.55 | 0.51 | 0.45 | 0.28 | 0.20 | 0.31 | -56 |
| PP2 | 0.72 | 0.70 | 0.69 | 0.71 | 0.67 | 0.57 | 0.56 | 0.42 | -42 |
| LP1 | 4.78 | 4.81 | 4.88 | 4.89 | 5.10 | 5.47 | 5.38 | 5.87 | +23 |
| LP2 | 6.16 | 6.67 | 6.89 | 7.02 | 7.38 | 8.55 | 10.28 | 11.26 | +83 |
| RC1 | 0.69 | 0.52 | 0.43 | 0.39 | 0.30 | 0.10 | 0.10 | 0 | -100 |
| RC2 | 1.62 | 1.54 | 1.49 | 1.50 | 1.30 | 1.32 | 1.35 | 1.08 | -33 |
| SF1 | 4.09 | 3.11 | 2.67 | 2.41 | 1.70 | 0.73 | 0.38 | 0.27 | -93 |
| SF2 | 8.01 | 7.90 | 7.81 | 7.71 | 7.44 | 6.98 | 6.71 | 5.43 | -32 |
| WBP | 10.64 | 11.72 | 12.14 | 12.37 | 13.26 | 14.28 | 15.18 | 16.77 | +58 |
| CL | 0.40 | 0.15 | 0.09 | 0.09 | 0.04 | 0 | 0 | 0 | -100 |

[a] Calculated from the difference between 2 ha and 400 ha percentages.

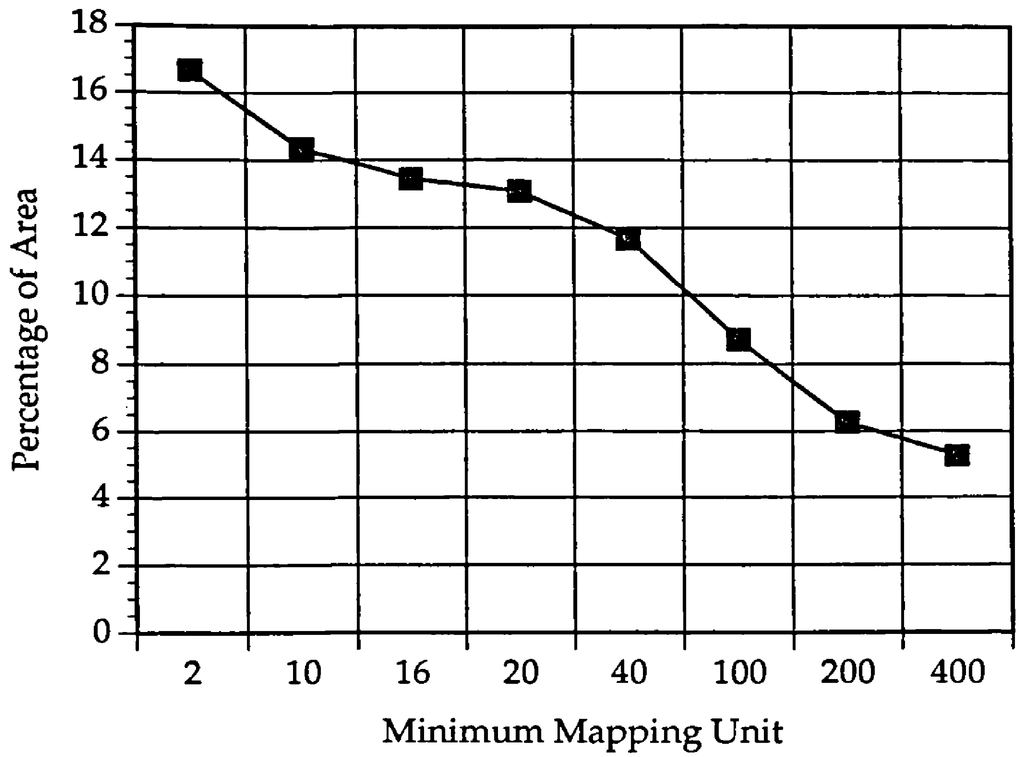


Figure 3-16. Decrease in area occupied by all cover types except conifers as minimum mapping unit increases.

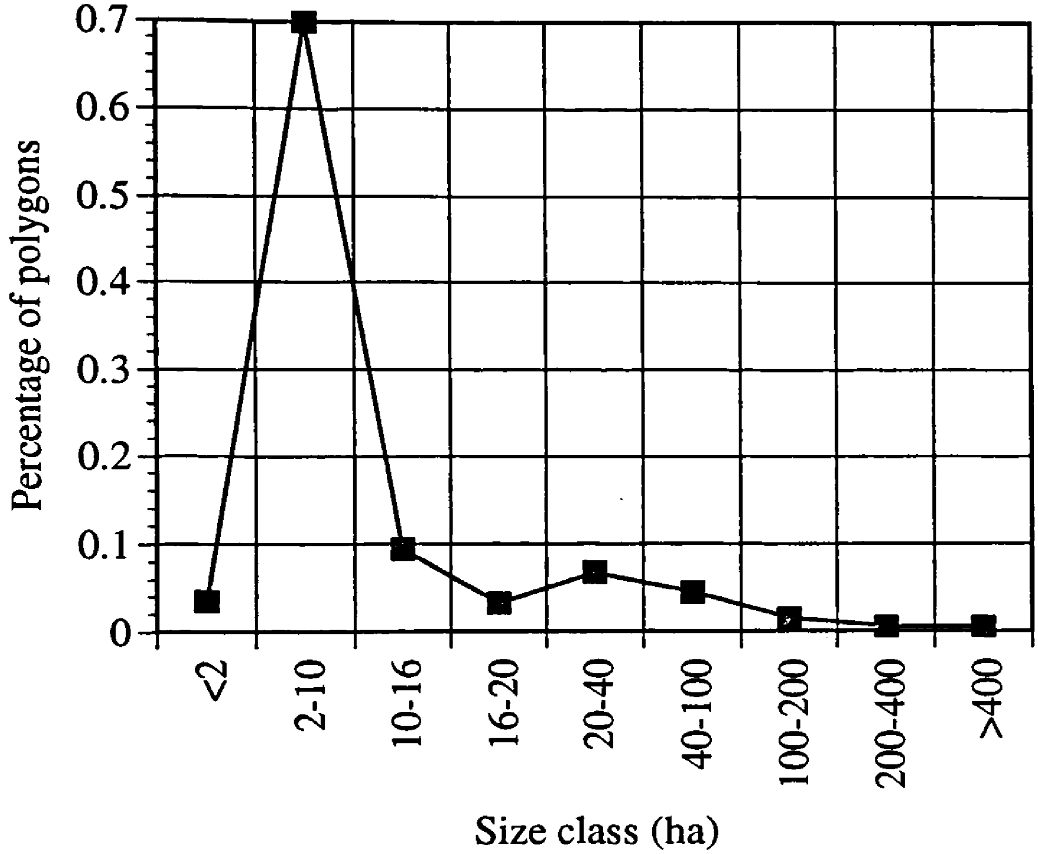


Figure 3-17. Frequency distribution of polygons by size class for the standardized vegetation layer, 2 ha MMU, Seeley-Swan landscape, northwestern Montana. At 2 ha MMU, the landscape includes 13,247 polygons.

Table 3-12. Distribution of polygons by size class (ha) for the standardized vegetation layer, 2 ha minimum mapping unit (MMU), Seeley-Swan landscape, northwestern Montana. Size classes were chosen to correspond to the MMUs used in the merging process; water was excluded from analysis.

| | | Percentage of polygons by size class (ha) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| CT[a] | NP[b] | <2 | <10 | <16 | <20 | <40 | <100 | <200 | <400 | <6000 |
| BAR | 348 | 16.1 | 60.3 | 6.3 | 2.3 | 7.5 | 4.0 | 1.4 | 0.9 | 1.2 |
| ROC | 389 | 8.2 | 75.3 | 5.9 | 2.3 | 5.4 | 2.6 | 0.3 | 0 | 0 |
| GRA | 725 | 1.1 | 80.3 | 9.5 | 1.2 | 4.7 | 2.8 | 0.4 | 0 | 0 |
| SHR | 1349 | 1.5 | 70.6 | 9.9 | 3.1 | 7.9 | 5.8 | 0.9 | 0.4 | 0 |
| B1 | 4 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| B2 | 9 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| URB | 2 | 50.0 | 0 | 0 | 50.0 | 0 | 0 | 0 | 0 | 0 |
| AGR | 85 | 0 | 54.1 | 16.5 | 11.8 | 5.9 | 9.4 | 2.4 | 0 | 0 |
| BU | 12 | 0 | 58.3 | 16.7 | 8.3 | 0 | 16.7 | 0 | 0 | 0 |
| **CUT** | **1696** | **2.7** | **62.2** | **10.4** | **4.1** | **8.6** | **6.7** | **3.0** | **1.0** | **1.3** |
| MC1 | 1306 | 3.9 | 70.5 | 9.0 | 3.5 | 5.8 | 4.8 | 1.8 | 0.5 | 0.4 |
| **MC2** | **1292** | **3.0** | **66.6** | **9.6** | **4.0** | **8.4** | **4.4** | **2.2** | **1.1** | **0.7** |
| DF1 | 429 | 3.0 | 80.2 | 9.6 | 2.1 | 3.3 | 1.2 | 0.7 | 0 | 0 |
| **DF2** | **646** | **2.3** | **74.6** | **11.5** | **3.3** | **4.8** | **3.1** | **0.5** | **0** | **0** |
| PP1 | 174 | 1.7 | 83.3 | 6.3 | 1.2 | 1.7 | 5.2 | 0 | 0.6 | 0 |
| PP2 | 134 | 0.7 | 73.1 | 5.2 | 2.2 | 10.4 | 6.0 | 2.2 | 0 | 0 |
| **LP1** | **676** | **3.0** | **73.2** | **8.6** | **4.0** | **5.9** | **3.3** | **1.0** | **0.4** | **0.6** |
| **LP2** | **673** | **6.8** | **67.0** | **9.7** | **2.5** | **7.6** | **3.7** | **1.6** | **0.3** | **0.7** |
| RC1 | 259 | 2.7 | 83.0 | 8.5 | 2.3 | 2.3 | 1.2 | 0 | 0 | 0 |
| RC2 | 298 | 3.7 | 71.5 | 7.1 | 5.0 | 8.1 | 2.0 | 2.0 | 0.7 | 0 |
| SF1 | 892 | 2.4 | 68.7 | 11.4 | 4.0 | 8.0 | 4.7 | 0.7 | 0.1 | 0 |
| SF2 | 825 | 6.8 | 60.8 | 8.6 | 3.8 | 7.9 | 5.7 | 3.9 | 2.2 | 0.4 |
| **WBP** | **670** | **21.9** | **45.7** | **6.3** | **3.1** | **8.7** | **7.3** | **2.8** | **2.5** | **1.6** |
| CL | 354 | 58.2 | 36.7 | 3.7 | 0.3 | 0.8 | 0.3 | 0 | 0 | 0 |
| All | 13247 | 6.0 | 67.4 | 9.1 | 3.3 | 6.8 | 4.5 | 1.6 | 0.7 | 0.5 |

[a] Cover types; those increasing in areal extent as MMU increases shown in bold.

[b] Number of polygons.

Table 3-13. A comparison of descriptive statistics at a variety of minimum mapping units for the Seeley-Swan landscape (247,925 ha), northwestern Montana. (Continued on following page.)

| LANDSCAPE MEASURE | MINIMUM MAPPING UNIT 2 ha | 10 ha | 16 ha | 20 ha |
|---|---|---|---|---|
| Number of patches | 13,247 | 3791 | 2609 | 2186 |
| Mean patch size (ha) | 18.44 | 64.45 | 93.65 | 111.77 |
| Patch size standard deviation (ha) | 102.71 | 236.62 | 328.29 | 379.07 |
| Largest patch index (%) | 2.41 | 3.20 | 4.48 | 4.59 |
| Mean shape index | 1.88 | 2.46 | 2.66 | 2.75 |
| Mean patch fractal dimension | 1.10 | 1.13 | 1.14 | 1.14 |
| Total edge (m) | 20,081,940 | 14,159,970 | 12,656,520 | 12,000,780 |
| Total core area (ha) | 105,253 | 137,729 | 147,377 | 151,705 |
| Number of core areas | 18,439 | 10,251 | 8076 | 7213 |
| Mean core area index (%) | 14.24 | 37.27 | 43.94 | 46.82 |
| Mean nearest neighbor (m) | 334.80 | 609.30 | 757.00 | 844.30 |
| Nearest neighbor standard deviation (m) | 828.14 | 1130.72 | 1579.71 | 1759.55 |
| Patch richness | 24 | 22 | 22 | 21 |
| Shannon's diversity index | 2.53 | 2.44 | 2.41 | 2.39 |
| Simpson's diversity index | 0.89 | 0.88 | 0.88 | 0.87 |
| Shannon's evenness index | 0.80 | 0.79 | 0.78 | 0.79 |
| Simpson's evenness index | 0.93 | 0.92 | 0.92 | 0.92 |
| Interspersion/juxtaposition index (%) | 74.55 | 74.31 | 73.17 | 74.04 |
| Contagion (%) | 50.00 | 52.67 | 53.93 | 53.82 |

Table 3-13 (continued). A comparison of descriptive statistics at a variety of minimum mapping units for the Seeley-Swan landscape (247,925 ha), northwestern Montana.

| LANDSCAPE MEASURE | MINIMUM MAPPING UNIT 40 ha | 100 ha | 200 ha | 400 ha | SPEARMAN RANK[a] |
|---|---|---|---|---|---|
| Number of patches | 1212 | 542 | 280 | 164 | -1.000 |
| Mean patch size (ha) | 201.59 | 450.79 | 872.61 | 1489.81 | 1.000 |
| Patch size standard deviation (ha) | 579.4 | 1239.11 | 2063.35 | 2891.43 | 1.000 |
| Largest patch index (%) | 4.89 | 8.75 | 9.57 | 10.01 | 1.000 |
| Mean shape index | 3.10 | 3.63 | 4.14 | 4.55 | 1.000 |
| Mean patch fractal dimension | 1.15 | 1.17 | 1.17 | 1.17 | 0.970 |
| Total edge (m) | 10,116,720 | 7,933,020 | 6,473,520 | 5,526,540 | -1.000 |
| Total core area (ha) | 164,583 | 180,176 | 190,855 | 197,895 | 1.000 |
| Number of core areas | 5092 | 3199 | 2319 | 1848 | -1.000 |
| Mean core area index (%) | 54.89 | 63.65 | 67.36 | 68.85 | 1.000 |
| Mean nearest neighbor (m) | 1058.20 | 1801.10 | 2625.70 | 2425.40 | 0.976 |
| Nearest neighbor standard deviation (m) | 1826.75 | 4381.35 | 6416.66 | 4100.36 | 0.929 |
| Patch richness | 21 | 20 | 18 | 15 | -0.988 |
| Shannon's diversity index | 2.33 | 2.20 | 2.11 | 2.06 | -1.000 |
| Simpson's diversity index | 0.86 | 0.85 | 0.83 | 0.83 | -0.988 |
| Shannon's evenness index | 0.77 | 0.73 | 0.73 | 0.76 | -0.880 |
| Simpson's evenness index | 0.91 | 0.89 | 0.88 | 0.89 | -0.933 |
| Interspersion/juxtaposition index (%) | 72.22 | 68.84 | 67.75 | 70.58 | -0.905 |
| Contagion (%) | 55.77 | 58.41 | 59.28 | 58.05 | 0.905 |

[a] Correlation between individual metrics and minimum mapping unit.

Table 3-14. Landscape metrics showing similar response to increasing minimum mapping unit (MMU).

| MONOTONIC INCREASE |
|---|
| mean patch size |
| patch size standard deviation |
| largest patch index |
| mean shape index |
| mean patch fractal dimension |
| total core area |
| mean core area index |
| mean nearest neighbor |
| nearest neighbor standard deviation |
| **MONOTONIC DECREASE** |
| number of patches |
| total edge |
| number of core areas |
| patch richness |
| Shannon's diversity |
| Simpson's diversity |
| Simpson's evenness |
| **INCONSISTENT TREND** |
| Shannon's evenness |
| interspersion/juxtaposition |
| contagion |

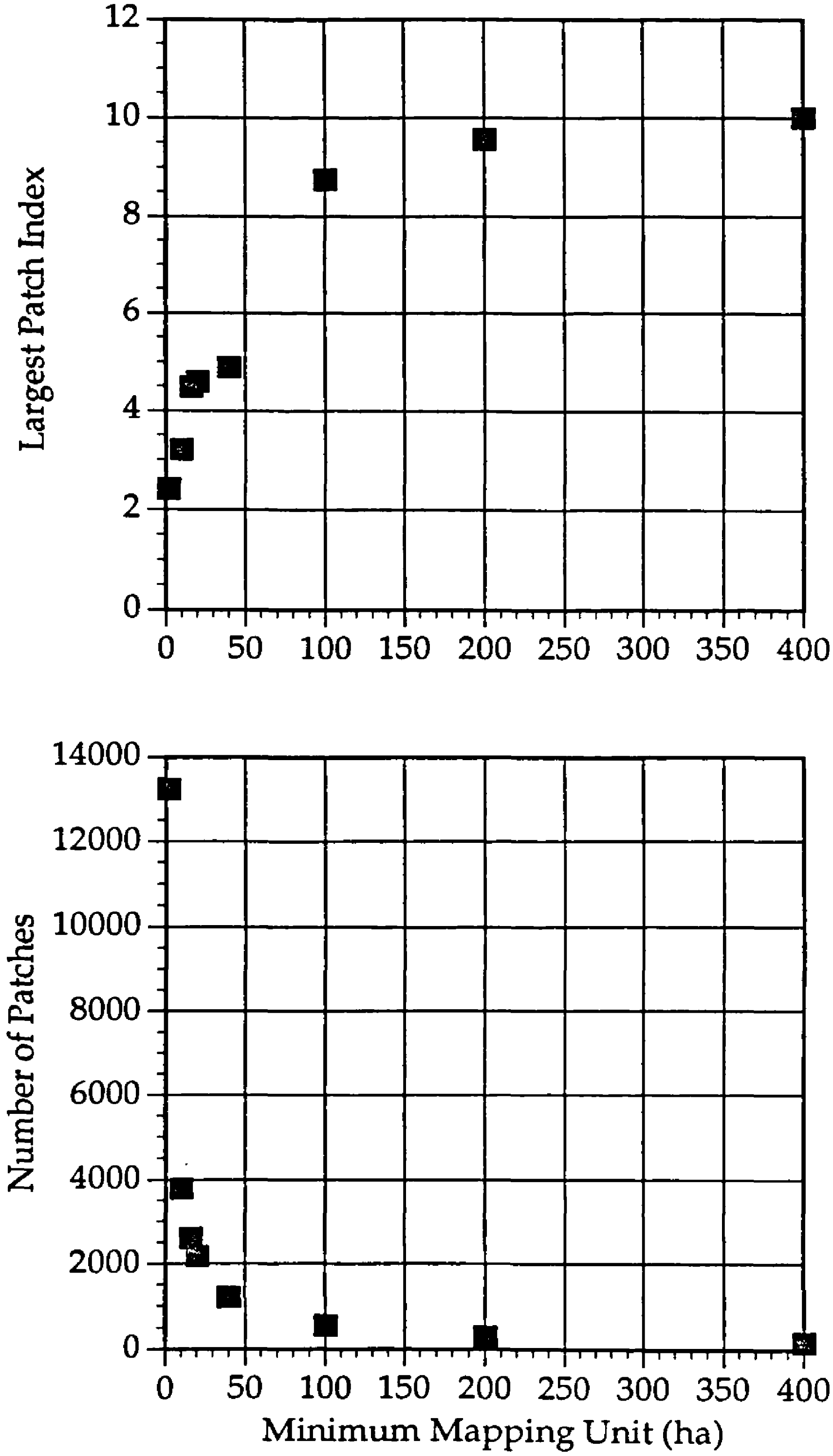


Figure 3-18. Relationship between selected landscape metrics and minimum mapping units (continued).

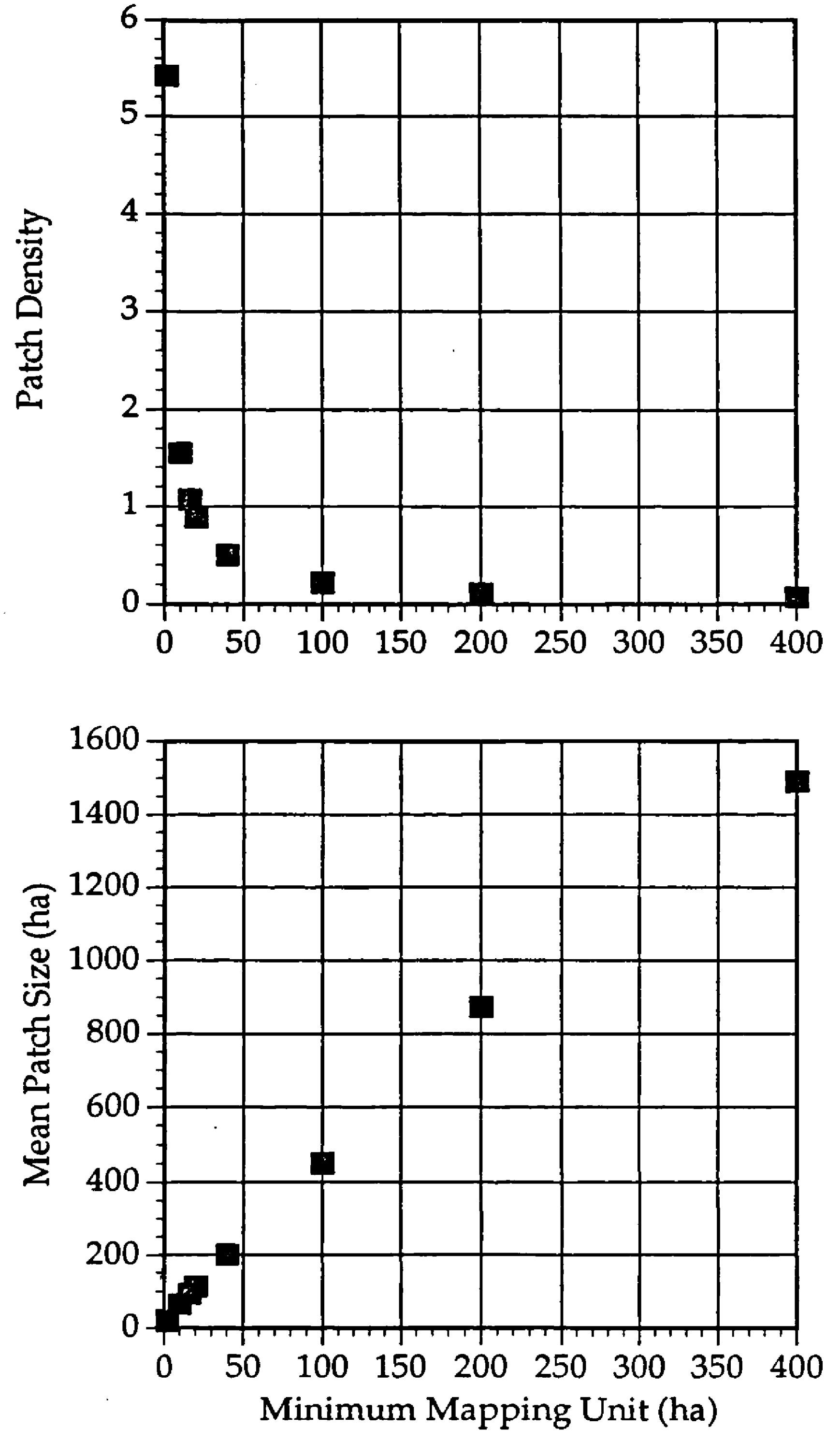


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units.

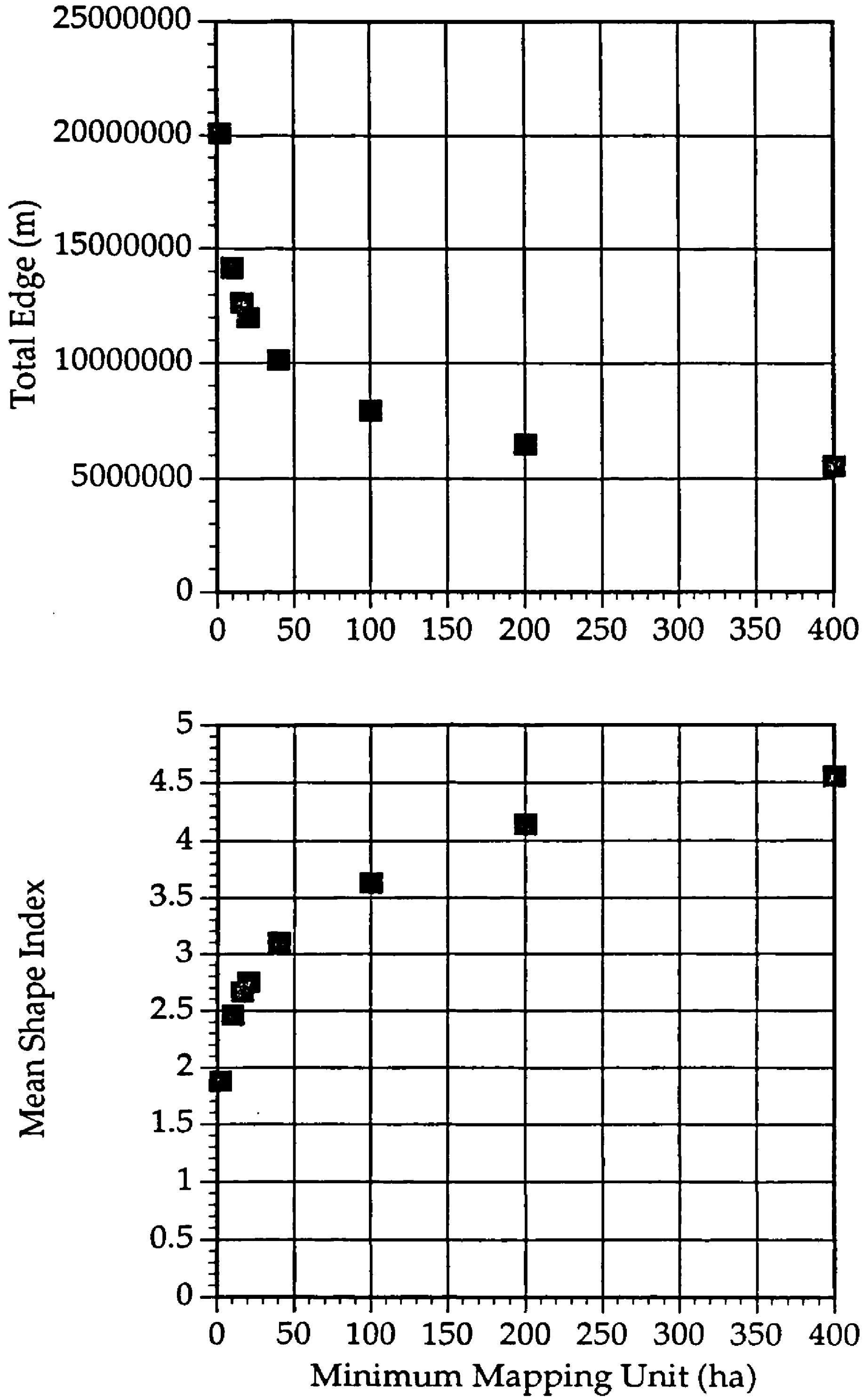


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units.

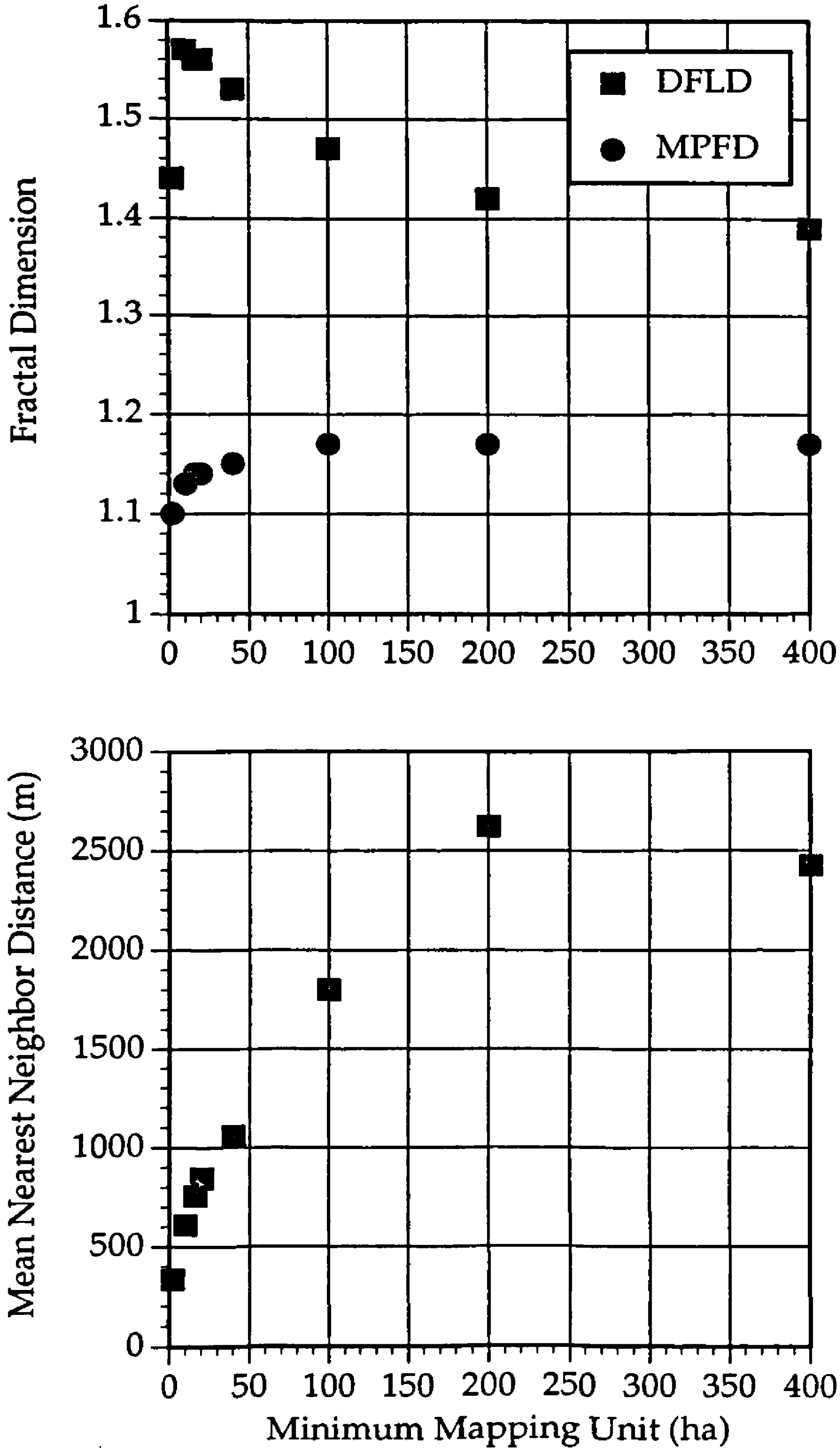


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units. DLFD = double log fractal dimension, MPFD = mean patch fractal dimension.

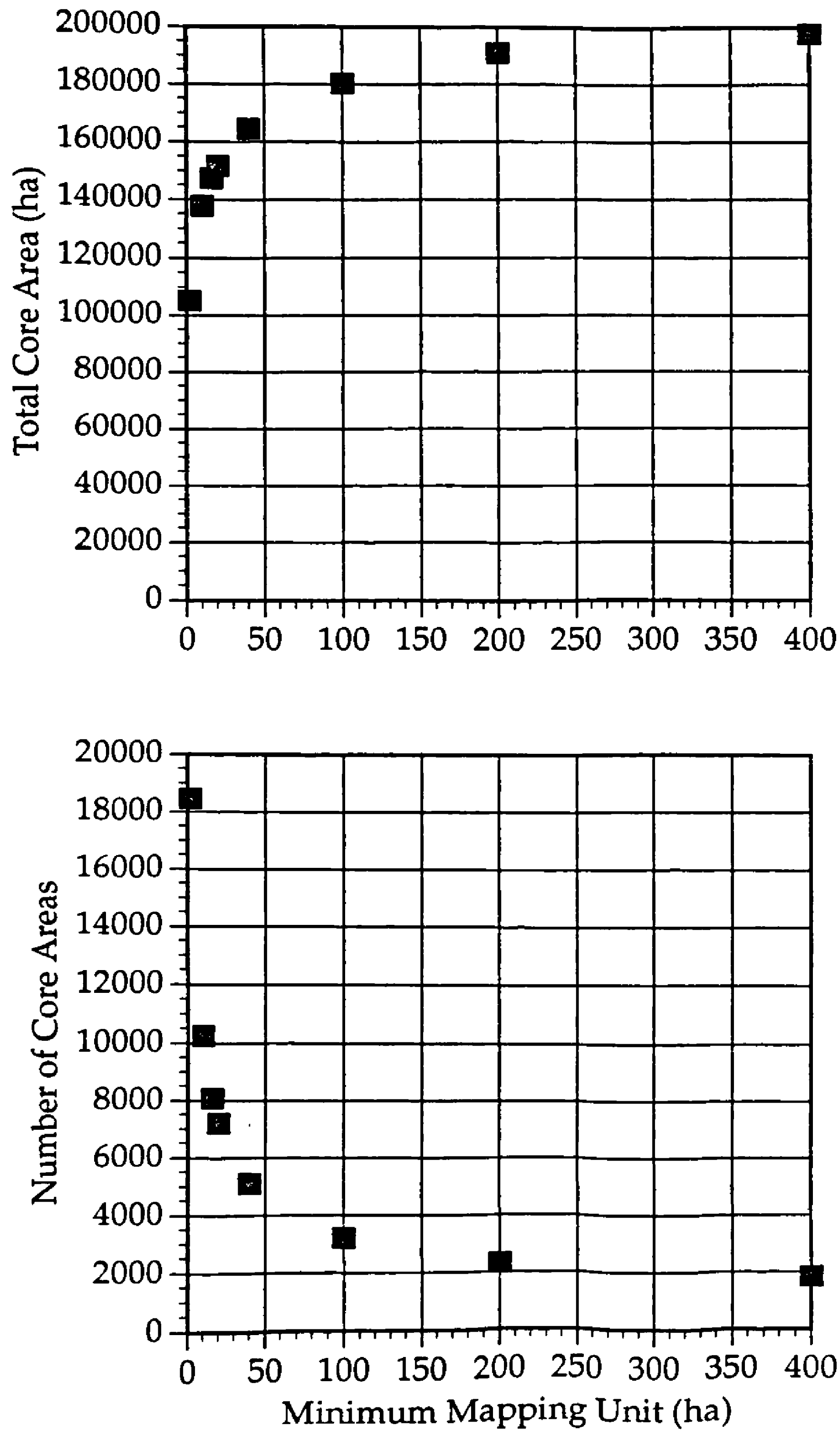


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units.

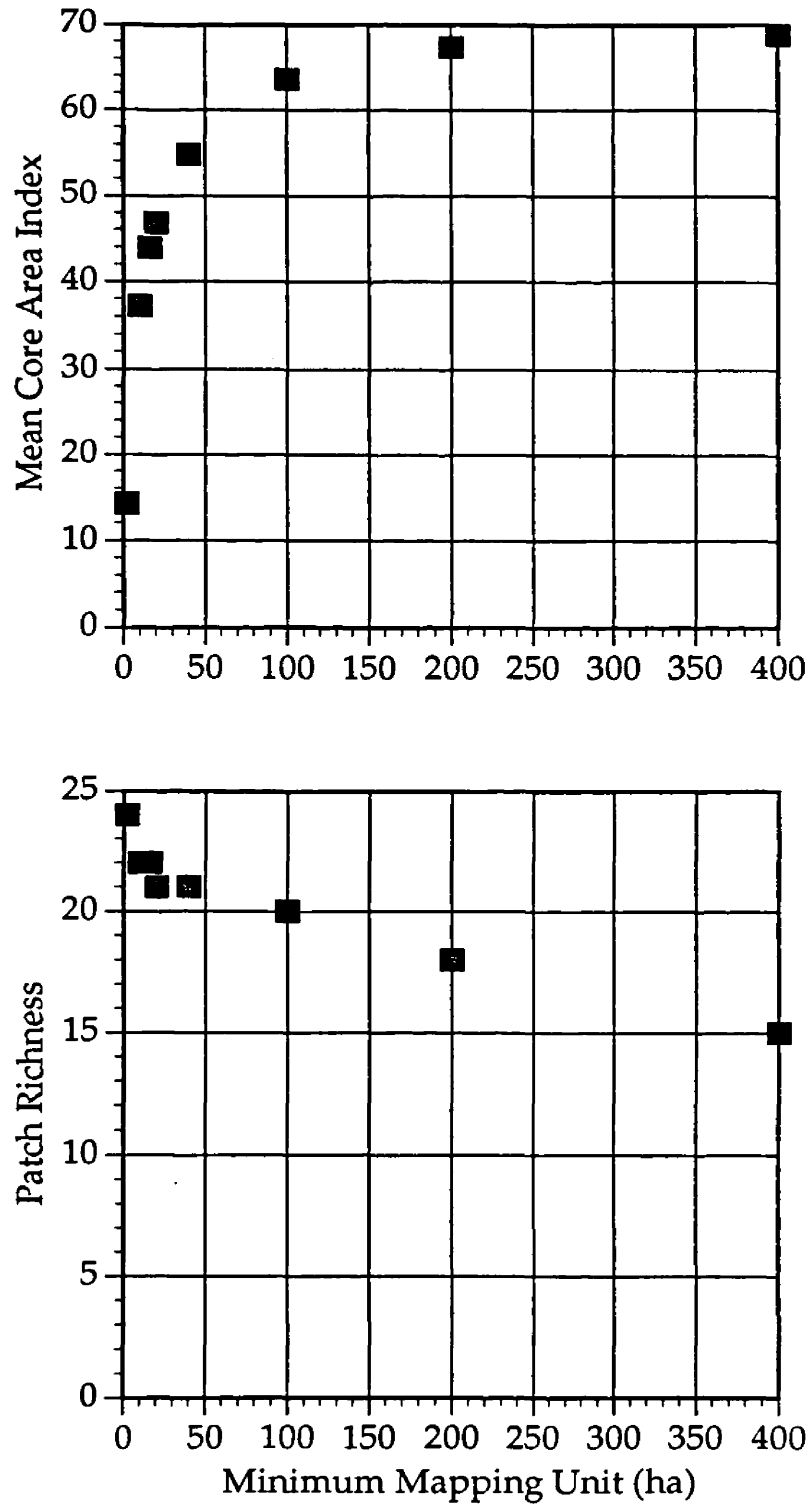


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units.

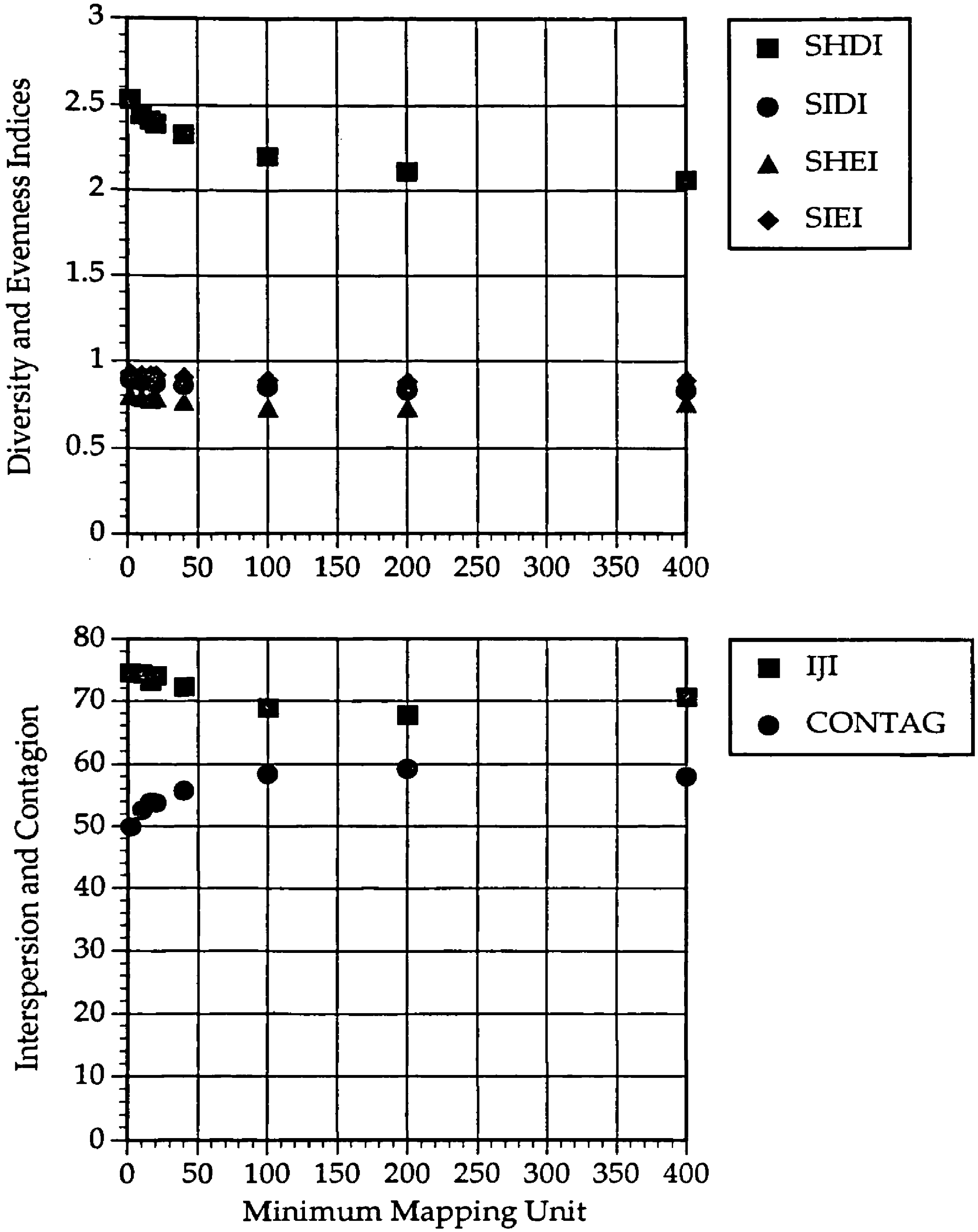


Figure 3-18 (continued). Relationship between selected landscape metrics and minimum mapping units. SHDI = Shannon's diversity, SIDI = Simpson's diversity, SHEI = Shannon's evenness, SIEI = Simpson's evenness, IJI = interspersion/juxtaposition, CONTAG = contagion.

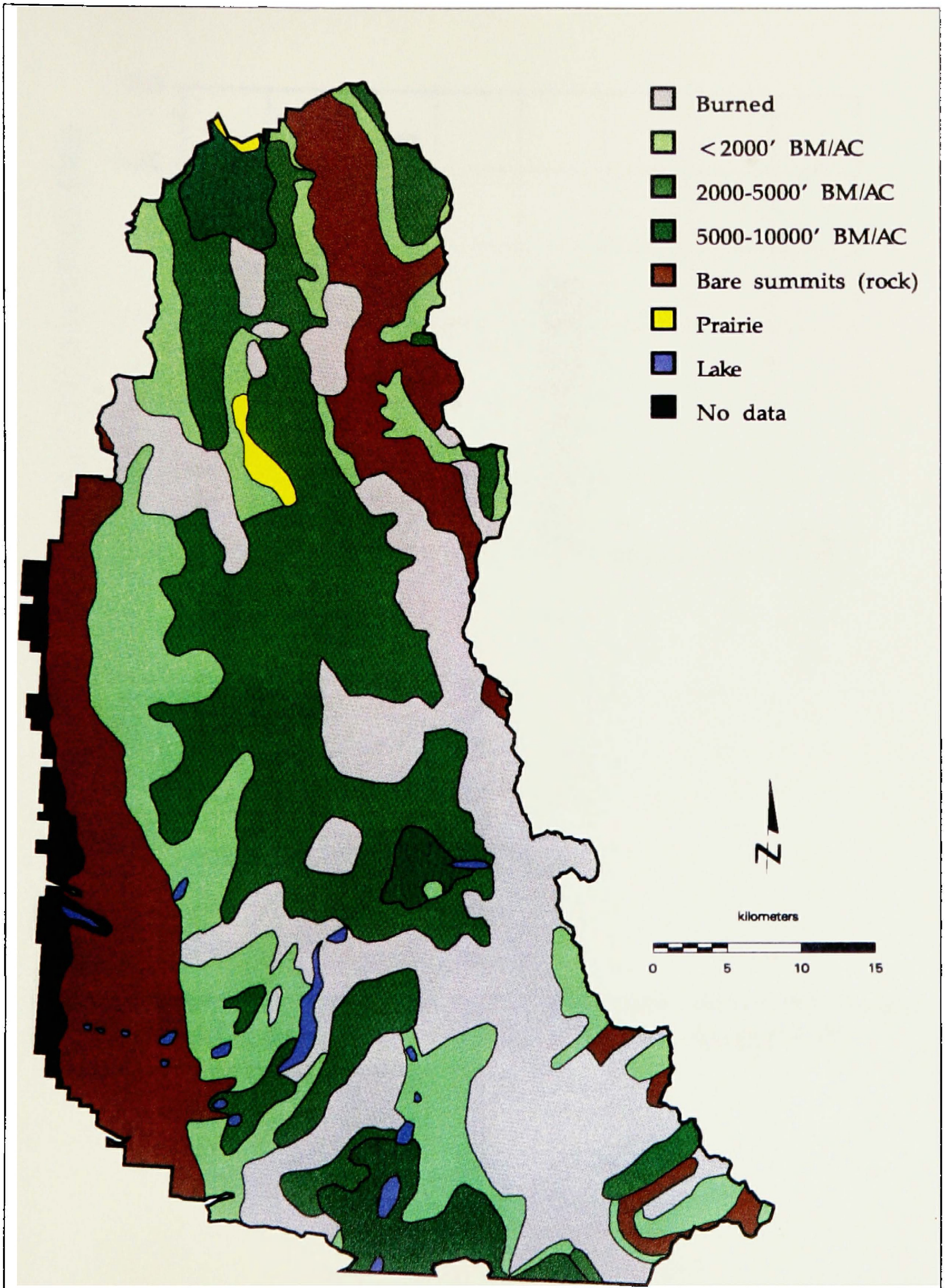


Figure 3-19. The Seeley-Swan landscape as mapped by H.B. Ayres (1900) as part of a survey of the Lewis and Clark Forest Reserve. Original scale roughly 1:500,000.

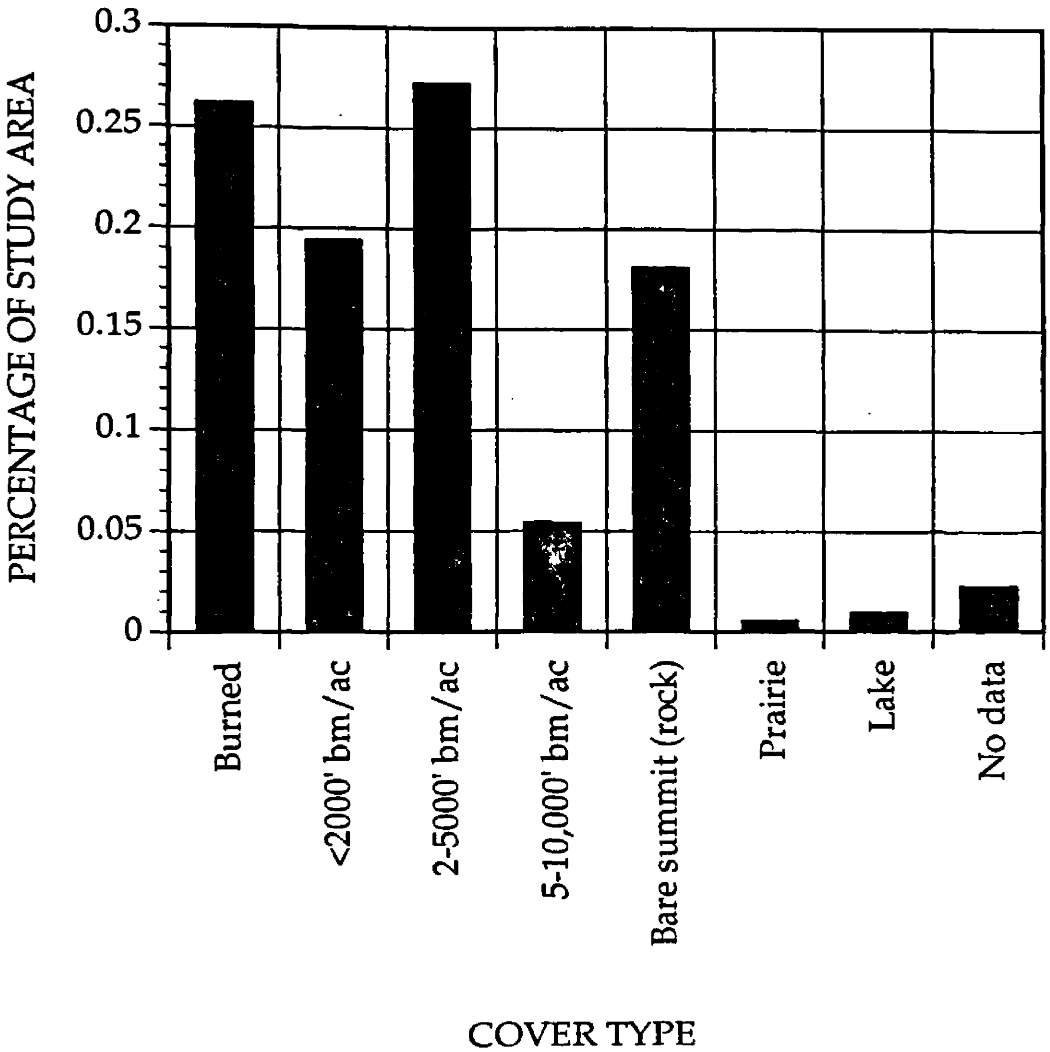


Figure 3-20. Distribution of cover types within the Seeley-Swan landscape in 1899 as mapped by H.B. Ayres (1900).

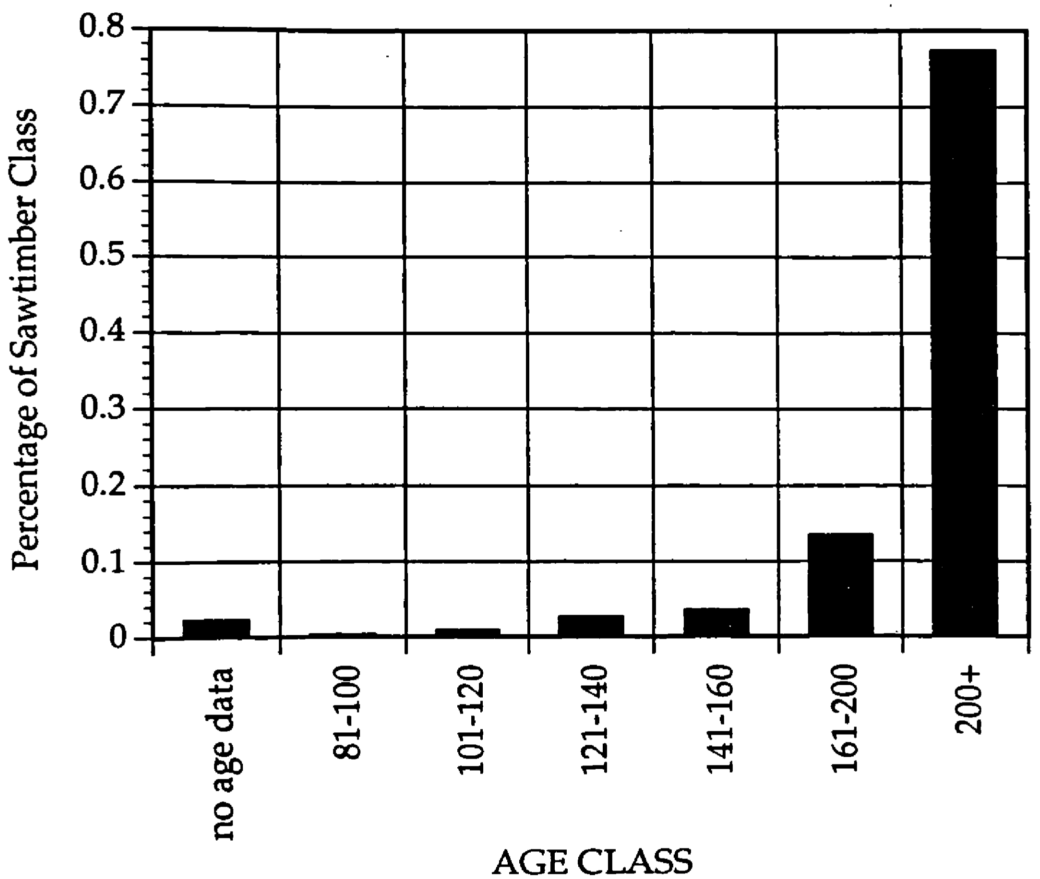


Figure 3-21. Distribution of age classes within the sawtimber stand class as mapped in the 1930s, Seeley-Swan landscape, northwestern Montana.

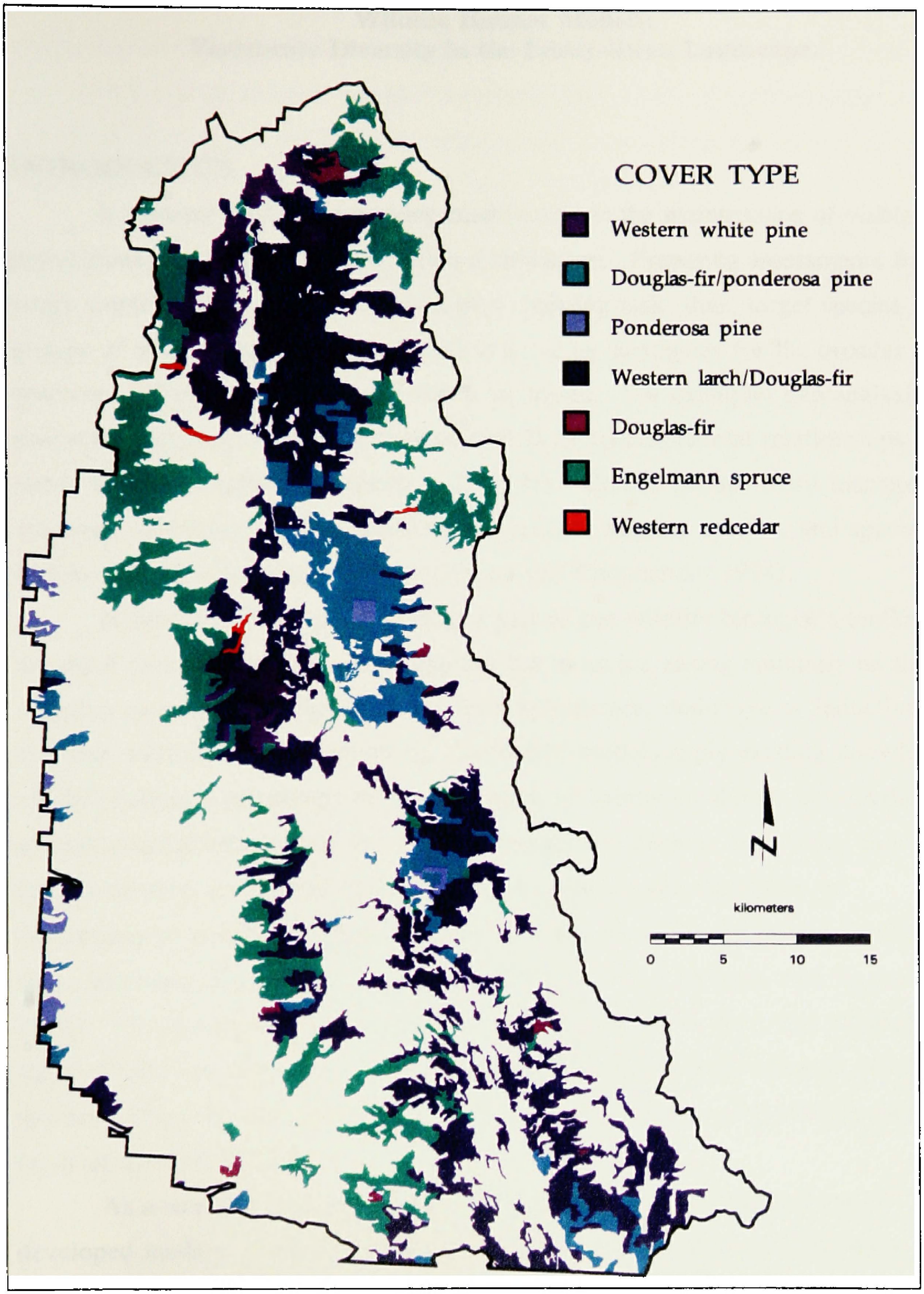


Figure 3-22. Predicted distribution of old-growth forest in the 1930s for the Seeley-Swan landscape, northwestern Montana.

**Chapter 4.**
**Wildlife Habitat Models:**
**Vertebrate Diversity in the Seeley-Swan Landscape.**

## INTRODUCTION

A primary goal of conserving biodiversity is the maintenance of viable populations of all native species within a landscape. Preparing assessments for every single species, however, would be a daunting task; thus, target species or groups of species are typically selected to serve as surrogates for the broader spectrum of biodiversity in conservation strategies. For example, gap analysis procedures involve the construction of models of species-habitat relationships for all native terrestrial vertebrates (Scott et al. 1993). Certain groups merit management emphasis, including keystone species, umbrella or flagship species, and species highly vulnerable to human activities (Noss and Cooperrider 1994).

Habitat modeling has long been a part of the wildlife biologist's toolkit, but the rapid evolution of GIS technology has led to an increasing emphasis on the mapping component of such models. Two approaches, deductive or inductive, may be taken with GIS habitat modeling. Deductive models apply existing knowledge of wildlife-habitat relationships to generate maps of habitat conditions and predicted species distributions; again, gap analysis exemplifies such an approach. Inductive models develop correlations between animal locations and vegetative or environmental conditions, then use these correlations to predict habitat conditions (e.g., see Agee et al. 1989, Aspinall and Veitch 1993, Lehmkuhl and Raphael 1993). The latter approach may yield more accurate predictions, but requires the input of data that simply has not been gathered for many landscapes, or even many species. Thus, the deductive approach proves more feasible for multiple-species analyses over broad areas.

As a means of visualizing vertebrate diversity in the Seeley-Swan, I developed models of wildlife-habitat relationships for 20 species (Table 4-1), then mapped predicted habitat for the 1930s and 1990s. Species were selected to represent a variety of taxa, habitats, and ecological roles. Heavy emphasis was placed on species accorded special management status, including species on the Forest Service Northern Region sensitive list (Jolly 1994) and Flathead NF management indicator species (USDA:FS 1985, 1992). Although all species are known to occur in the general vicinity, some have not been documented within the study area itself. Actual occurrence data are scanty, thus precluding either an inductive modeling approach or a thorough validation of habitat predictions. The resultant maps portray how wildlife distributions may have changed in the landscape over time, based on current knowledge and assumptions about wildlife-habitat relationships.

## METHODS

### Road Density

Because road densities may be important indicators of habitat quality for species sensitive to human disturbance (i.e., gray wolf and grizzly bear), I mapped total road density ($mi/mi^2$) for the Seeley-Swan. Vector layers showing all roads and trails within the study area were acquired from Flathead and Lolo NFs, modified to eliminate trails, and converted to raster format. Road densities were mapped using a moving-circle technique called FOCALSUM (ESRI 1991) to calculate total road density for the square mile surrounding each 30 m cell in the study area. I then calculated the percentage of the landscape in road density classes ranging from 0 $mi/mi^2$ to 11.1-12.0 $mi/mi^2$. Details are described in Appendix B. Calculations were based on square miles rather than kilometers because the former measurement has become a standard in the wildlife literature, and the moving-circle results cannot readily be converted from metric to English units.

Using the same methodology, I illustrated changes in road density over time in a 143 $km^2$ section of the study area. This section, located in the central Swan Valley, was selected for analysis because a 1:45,256 scale map of roads had already been drafted from 1934 aerial photographs (Freedman and Habeck 1985). I digitized this map, converted it to raster format, and calculated road densities. I also digitized the boundary of the map and used it to clip out a portion of the 1994 road layer. Although I could have used the results of the previous analysis, values in this smaller area would have been influenced by roads outside the boundary of the 1934 map, thus inflating the 1994 results in comparison with those for 1934, where information on roads outside the map boundary was lacking. To avoid this, I did a separate moving-window analysis for this subset of the 1994 road layer.

**Wildlife Habitat**

I compiled information on habitat selection for each of 20 selected species from the existing literature, then drafted modeling rules for prediction of essential habitat (defined as that which is critical to the species' persistence in an area) based on GIS layers and attributes (Appendix C). Because information was limited for many species and the GIS attributes available did not always match habitat requirements, arbitrary decisions sometimes were made during model construction. I illustrated the sensitivity of the results to modeling definitions by varying the parameters for lynx denning habitat using the 1930s and 1990s vegetation layers.

I wrote programs in Arc Macro Language (AML) to create three layers of predicted habitat for each species: one for the 1930s (16 ha MMU), and two for the 1990s (2 ha and 16 ha MMU). Appendix D provides a sample program for the peregrine falcon. My intent was to draw comparisons between the 1930s and 1990s at a 16 ha MMU; in addition, the 1990s data was processed at 2 ha MMU to provide input for the reserve selection process (Chapter 5), and to illustrate differences in predicted habitat for the 1990s as MMU was increased. Habitat layers were based primarily on vegetation layers and associated attributes, including standardized cover type and size class and mean elevation, slope, and aspect for each polygon (see Chapter 3 for details of construction). However, other layers were frequently incorporated, including layers for topography, hydrography, and road density. My objective was to predict the presence or absence of habitat for each species for each polygon in the vegetation layers -- I did not want to create completely new polygons in the modeling process and find myself unable to relate the results back to the original files, nor did I want to overestimate habitat by including entire polygons when only portions met the modeling rules. The most feasible solution proved to be construction and maintenance of separate "master" wildlife databases relating to each of the three vegetation layers. As the final step in the AML programs for each species, results of the habitat modeling process were written to each master database. Each polygon was assigned a 1/0 value for presence/absence of habitat. In addition, the proportion of the polygon estimated to be habitat was recorded after overlaying predicted habitat with original polygon boundaries (proportions were only $<1$ in situations where buffering techniques were used). This method maintained consistency and also allowed flexibility.

To quantify changes in habitat over time, habitat layers for the 1930s and 1990s at 16 ha MMU were converted from ARC/INFO GRID to ERDAS GIS format, then processed using FRAGSTATS (McGarigal and Marks 1994). A subset of spatial statistics was selected for interpretation, including number of patches, largest patch index, mean patch size and standard deviation, and mean nearest neighbor distance and standard deviation. Returning to the three master wildlife databases, I summed the 1/0 values for each species to obtain a count for each polygon, providing a traditional measure of species richness in the Seeley-Swan landscape based on my limited subset of 20 species.

## Scale

As a simple accompaniment to the discussion of spatial resolution in Chapter 3, I created habitat layers for the pileated woodpecker at eight MMUs: 2, 10, 16, 20, 40, 100, 200, and 400 ha. Modeling rules were identical to those presented in Appendix C. The pileated woodpecker was selected because its habitat model was based completely on vegetation. Incorporation of other data layers could confound the results by introducing potential methodological problems at increasing scales (e.g., mean topographic values becoming less meaningful because they represent larger areas). By avoiding the inclusion of other data layers, results solely reflect changes in MMU.

## RESULTS AND DISCUSSION

### Road Density

In the Seeley-Swan landscape, road densities in 1994 ranged from 0 mi/mi$^2$ to 11.1-12.0 mi/mi$^2$ (Fig. 4-1). Road densities are highest in the valley bottoms where human activities are concentrated, but this influence extends even to the edges of the roadless/wilderness areas at higher elevations (Fig. 4-2). About 41% of the Seeley-Swan is unroaded (0 mi/mi$^2$) and 6% of the area has road density $\leq 1$ mi/mi$^2$.

Not surprisingly, road densities have increased over the past 60 years in the central Swan Valley (Fig. 4-3). As the road network has become more complex, the percentage of area with road density greater than zero has increased dramatically, from 44.3% in 1934 to 99.7% in 1994. Not only has the area affected to some degree by roads increased, the actual road density values have increased as well: In 1934 the maximum road density was 3 mi/mi$^2$, while in 1994 the maximum value was 8 mi/mi$^2$ in the central Swan Valley (Fig. 4-4).

Calculation of road densities is an important part of evaluating wildlife habitat; as road densities increase, many species become more vulnerable to human-caused mortality. Thus, assessment of security areas is critical for species like the grizzly bear and wolf, as well as lynx, marten, fisher, bald eagle, and various game species. Road densities have obviously increased in the past 60 years in the Seeley-Swan as human settlements have expanded and timber harvest activities accelerated. Today, very little of the valley bottoms remains uninfluenced by roads; instead, the lower elevations are dominated by high road densities. This does not necessarily mean that grizzlies, wolves, and other animals avoid these areas, but that these animals will be less secure in these heavily roaded zones.

Admittedly, this is only a partial analysis of road density; a more complete assessment would distinguish between open and closed roads, and perhaps include trails as well. I was unable to create an open road layer for the entire study area because the Seeley Lake Ranger District portion lacked codes for closure status. However, I examined the Swan Lake Ranger District portion of the study area, where codes were available. In comparison with the analysis including all roads (as described above), I found much larger tracts of valley bottom to be unaffected by open roads, suggesting somewhat higher security for wildlife. I excluded trails from analysis because the primary reason for creating a road density layer was as input to the grizzly bear habitat model, where I used a cutoff value of 2 mi/mi$^2$ (Mace and Manley 1993). Trails were not included in the road density calculations which led Mace and Manley (1993) to arrive at this value; thus, I opted to exclude them from my calculations as well. Further research suggests that the degree of avoidance by grizzly bears differs between trails and roads (Kasworm and Manley 1990). Nevertheless, trails unquestionably improve accessibility to remote areas; certainly the major pack trails, especially those providing access to the Bob Marshall Wilderness, increase the probability of human-bear conflicts. If trails had been included, the higher elevations would be striped with road densities $>0$ winding up many drainages, but seldom would the cutoff value of 2 mi/mi$^2$ for grizzly bears be exceeded.

In addition to importance as an indicator of wildlife vulnerability, road density calculation is also a useful tool in assessing the "naturalness" of an area (*sensu* Margules and Usher 1981). Thus, total road densities will be used in designing a reserve network for the Seeley-Swan landscape (Chapter 5). Note that for this purpose, calculation of total rather than open road density is entirely appropriate -- whether or not a road is receiving use, its presence is a detriment to the surrounding area's natural qualities. Despite a variety of road closure types that may be implemented, a road is a road is a road, with apologies to Gertrude Stein.

**Wildlife Habitat**

As expected, both the number of patches and hectares of predicted habitat for lynx in the 1930s and 1990s changed markedly as rule definitions were varied (Table 4-2). Naturally, broad definitions of habitat parameters led to inclusion of more patches and prediction of more hectares of habitat, whereas narrow definitions had the opposite effect. In addition, the means of implementing the rules can significantly affect the results, as is evident for mean and majority aspect calculations. Because information on habitat requirements may be severely limited -- in this case, a sample of only four denning sites (Koehler 1990) was available for interpretation -- care must be taken when applying highly specific criteria. An iterative approach to habitat modeling thus proves useful, allowing exploration of the effects of various rules on amount and distribution of mapped habitat.

As demonstrated in Table 4-2, the results obtained through the habitat modeling process depend entirely on the definitions of habitat, which in turn depend on the availability of pertinent literature and appropriate GIS data. In defining modeling rules, I assigned greater importance to studies conducted nearest the Seeley-Swan and in environments most similar to the northern Rocky Mountains. I also examined the literature for trends in habitat use, and frequently placed more weight on patterns evident in multiple studies than on parameters identified in a single study. This is not to say that isolated studies were discounted; on the contrary, they were often heavily relied upon by necessity. Sample size, intensity, and methodology were also considered. For example, presence/absence data were typically accorded less weight than reproductive site locations. I was also forced to make fairly liberal translations of habitat parameters in the literature into parameters available through manipulation of the GIS database. As stated earlier, my goal was to map "essential" habitat, areas critical to a species' ability to make a living within the study area, based on my interpretation of the literature. In most cases, I did not attempt to exclude habitat patches based on their proximity to other patches, but rather to simply present the distribution of habitat based on my rules and assumptions. Assessments of quality are more appropriately left to future management applications.

Hectares of predicted habitat decreased for 16 of 20 species between the 1930s and 1990s (Table 4-3). Increases were predicted for the common loon, northern goshawk, peregrine falcon, and Shiras moose. The sharpest declines ($\geq$50%) were predicted for the harlequin duck, flammulated owl, bald eagle, gray wolf, and grizzly bear.

The number of patches of predicted habitat increased for 19 species between the 1930s and 1990s; the only exception was the flammulated owl (Table 4-4). Note that patches here represent continuous areas of predicted habitat, and include aggregations of polygons from the base vegetation layers. For 17 species, the largest patch index decreased, indicating that the single largest patch of habitat made up a smaller percentage of the total predicted habitat in the 1990s than in the 1930s. Mean patch size decreased for 19 species, and remained constant between the two time periods for the common loon. Mean nearest-neighbor distance decreased for 16 species, indicating that habitat patches were more closely spaced in the 1990s than in the 1930s. As a rule, standard deviations were much larger than means for patch size and nearest-neighbor distance, denoting high patch variability. Changes in the spatial distribution of predicted habitat over time are portrayed for each species in Figures 4-5 through 4-24.

<u>Tailed Frog.</u> Perennial, fairly high-gradient streams with forested cover appear to be the limiting factor in the distribution of tailed frogs, assuming that all of the species' needs could be met within such areas. Total hectares of predicted habitat decreased by about 34% between the 1930s and 1990s. Because the only GIS layer that differs between the two models is vegetation, this decline can be attributed to a decrease in pole-sized or larger forest cover adjacent to perennial streams. Not only is less habitat predicted to be present in today's landscape, its distribution appears to be more fragmented as well: While the number of habitat patches nearly doubled, mean patch size decreased. In both time periods, predicted

habitat is fairly evenly dispersed throughout the landscape.

Harlequin Duck. Harlequin ducks were not easy to model because their habitat requirements are not fully understood, and many seemingly important parameters are not found in a typical GIS database. Examples include the availability of loafing sites, stream substrate, water quality, and sediment loading (Kuchel 1977, Wallen 1987, Cassirer and Groves 1991). Nevertheless, I designed a basic model assuming that low-gradient perennial streams with either shrub, broadleaf, or mature/overmature forest vegetation types were important for harlequin ducks. The amount of predicted habitat was quite small, and was restricted primarily to sections of the Swan and Clearwater Rivers. In the 1990s, about 56% less habitat was predicted to occur than in the 1930s, and its distribution seemed to be less contiguous, occurring in more numerous but smaller patches. The limited amount of predicted habitat, even under such a generalized model, accords well with the failure to document presence of harlequin ducks within the study area (Carlson 1990).

Common Loon. As is true for the harlequin duck, some important habitat parameters for the loon are not often incorporated in GIS databases; examples include water clarity and fluctuations in water level (Fitch 1989). I identified lakes $\geq$4 ha with at least 25% of the shoreline in pole-sized or larger forest (not necessarily contiguous) as loon habitat. Results were nearly identical for the two time periods in terms of area, number of patches, and mean patch size. However, different lakes were selected in the 1930s and 1990s, reflecting differences in percentage of shorelines that were forested.

Townsend's Warbler. Habitat for Townsend's warbler was predicted to be found in mature/overmature mixed conifer, Douglas-fir, western red cedar, and Engelmann spruce/subalpine fir stands. The area of predicted habitat declined by 31% between the 1930s and 1990s, and became increasingly fragmented as well.

Black-backed Woodpecker. I assumed that recently-burned areas were the most critical habitat component for black-backed woodpeckers; their occurrence in studies of burned areas is about 80%, versus 5-10% in general studies (Hutto, pers. comm.). However, I also included several mature/overmature forest types comprising the vast majority of predicted habitat. Overall, predicted habitat declined 29% between the 1930s and 1990s; almost certainly more significant is the reduction in area of recent burns (from 4439 ha to 149 ha). In the 1930s, fairly sizeable burns were scattered throughout the landscape, whereas only a few small burns are present today. This almost complete loss of a high-quality habitat component may represent a significant impact on black-backed woodpecker populations, especially when coupled with a general reduction in area of mature/overmature forest.

Pileated Woodpecker. I assumed that nesting habitat was the limiting factor for pileated woodpeckers, and that foraging areas would be adequately represented in a model of nesting habitat. In the model, I selected mature/overmature broadleaf, mixed conifer, Douglas-fir, ponderosa pine, western red cedar, and Engelmann spruce/subalpine fir forest types. Inclusion of high stand density as a selection criterion would probably have improved the model, but would not have allowed for comparison between the two time periods. As modeled, the amount of habitat declined 40% between the 1930s and 1990s; habitat loss occurred mostly at lower elevations. Predicted habitat also became less contiguous; whereas the largest patch made up over 40% of the total predicted area in the 1930s, it occupied less than 5% of the total area in the 1990s. Mean patch size also declined markedly.

Flammulated Owl. In this model, I simply selected the mature/overmature ponderosa pine stands with which the species has been associated throughout the northern and central Rocky Mountains (Bull and Anderson 1978, Goggans 1986, Holt and Hillis 1987, Howie and Ritcey 1987, Reynolds and Linkhart 1987, Bull et al. 1990, Reynolds and Linkhart 1992). An ability to select stands with low or moderate density would likely have strengthened the model. As predicted, amount of habitat declined by 88% between the 1930s and 1990s. Predicted habitat was not extensive in the 1930s, but it was concentrated in a few large patches scattered throughout the lower elevations. By the 1990s, most of the predicted habitat occurred in small remnant stands in the central Swan Valley. The flammulated owl is the only species for which the number of patches actually declined over time; largest patch index also decreased, along with mean patch size. However, the mean distance between habitat patches increased by nearly half a kilometer. Examination of the spatial arrangement of predicted habitat and related patch statistics suggests significant fragmentation and loss of flammulated owl habitat since the 1930s.

Boreal Owl. As a model of nesting habitat for the boreal owl, I selected for mature/overmature mixed conifer, Douglas-fir, Engelmann spruce/subalpine fir, and broadleaf forest types at or above 1300 m. In modeling nesting habitat, prime areas for roosting and foraging (mature/overmature spruce-fir, Hayward et al. 1993) are also included. The amount of predicted habitat decreased 16% between the 1930s and 1990s; this decline is perhaps not as sharp as observed for other species finding habitat in mature/overmature forests because boreal owls are restricted to higher elevations. As with most of the species modeled, the landscape for boreal owls has become increasingly fragmented over the past 60 years.

Barred Owl. The barred owl, having recently expanded its range into western North America (Shea 1974, Taylor and Forsman 1976, Boxall and Stepney 1982), appears to be somewhat of a generalist, although habitat preferences in the West have not been well studied. I selected all mature/overmature broadleaf and coniferous forest types $\leq$1800 m for inclusion in this habitat model, finding a 30% decrease in the amount of predicted habitat between the 1930s and 1990s. Again, predicted habitat appears to be more fragmented today than in the 1930s, but the barred owl's ability to colonize a wide variety of habitats (as seen in British Columbia, Dunbar et al. 1991) suggests a fairly broad environmental tolerance. This model might be improved by consideration of riparian corridors, which the barred owl appears to favor.

Northern Goshawk. The habitat model for the northern goshawk was one of the most restrictive that I designed, and could have been even more so had I included high stand density, which appears to be a preferred habitat characteristic (Reynolds et al. 1982, Crocker-Bedford and Chaney 1986, Hayward and Escano 1989, Whitford 1991). I focused on nesting habitat, selecting all mature/overmature coniferous forest types on gentler slopes (mean $\leq$40%) and northerly aspects (0-45° and 315-360°). Initially, I had difficulties with the aspect criterion because I was using mean values per polygon. As a result of the measurement's circular nature, few polygons have mean northerly aspects (e.g., the means of 170 and 190 *and* 1 and 359 are both 180). I circumvented this difficulty by creating a layer of northerly aspects only from the DEM, overlaying it with the vegetation layer, and calculating the percentage of each vegetative polygon occupied by northerly aspects. I then selected for "majority" aspect -- polygons with at least 50% northerly aspect. Even with this modification, only a limited amount of habitat was predicted. Interestingly, this was one of a handful of models where the amount of predicted habitat for the 1990s was much greater at 2 ha than at 16 ha MMU. This seems logical because larger polygons would be expected to contain a wider variety of aspects than smaller ones, and thus would be less likely to satisfy the majority criterion. This prediction can be generalized to apply to all techniques that rely on overlaying two attributes and calculating the percentage of one in the other. It also perhaps explains the 44% increase in amount of predicted habitat between the 1930s and 1990s: Although both periods were mapped at 16 ha MMU, the mean patch size is 153 ha for the 1930s and 72 ha for the 1990s. The larger polygons of the 1930s may have been less likely to meet the majority aspect criterion. I search for an explanation in my GIS layers and methodology because it seems implausible to me that, given a general decrease in areal extent of mature/overmature forest types over time, such types would be more likely to occur on gentler slopes and north aspects in the 1990s than the 1930s. Regardless of the reasons, predicted habitat in the Seeley-Swan landscape is quite limited and widely scattered. It is very possible that the gentle slopes and north aspects used by goshawks, as well as the preference for dense stands, reflect selection for cool, moist microclimates (Reynolds et al.

1982). Other stands lacking these topographical characteristics may still be able to provide the required environment, especially in the cool, moist Seeley-Swan, and thus there may be a greater amount of suitable habitat than predicted. Nevertheless, based on habitat parameters identified in studies to date, habitat appears to be limited in this landscape currently, and to have been historically limited as well.

Bald Eagle. I predicted that nesting habitat for bald eagles would be found in mature/overmature broadleaf, mixed conifer, Douglas-fir, and ponderosa pine types at elevations $\leq 1385$ m, as long as the stands fell at least partially within 1610 m of selected water bodies. (Water features within the 1610 m buffer were also included in the model as foraging habitat.) Several other factors could have been included in the model, including stand size, line-of-sight view to an associated water body, and distance to open roads (MBEWG 1991). I felt that stand size was not important given an MMU of 16 ha. Distance to open roads could not be calculated because of incomplete digital data. Line-of-sight to an associated body of water is certainly an important criterion, but I was unable to adequately assess it using existing data in a GIS. Between the 1930s and 1990s, the amount of predicted habitat declined by 74%. This large difference can be partially explained by methodology; in most models, if a polygon fell at least partially within the selected zone, the entire polygon was counted as habitat. I have already noted that polygons tended to be larger for the 1930s than the 1990s, even at the same MMU. This tendency seems especially strong in the valley bottom, where forests have become increasingly fragmented by timber harvest over the past 60 years. Thus, the predicted habitat for the 1930s sprawls beyond the buffer, while predicted habitat for the 1990s is more closely confined to the buffer zone. To ensure that the reported decline was not entirely due to this difference, I examined the amount of predicted habitat within the buffer itself. For the 1930s, this figure was 16,420 ha, and for the 1990s, 5562 ha, indicating an overall decrease of 66%. Accompanying this decrease in area was an overall increase in habitat fragmentation as evidenced by changes in the number of habitat patches, their mean size, and the percentage of total habitat occupied by the largest patch.

Peregrine Falcon. In constructing this model, I assumed that nesting habitat (cliffs) was the limiting factor, but also assessed foraging habitat, recognizing the need for an adequate prey base in an eyrie's vicinity. I attempted to identify cliffs using the DEM, then selected for foraging habitat (grass, agricultural, and water types) within 16.1 km of cliffs. Nesting habitat remained constant between the two time periods because I used the same digital elevation model. However, foraging habitat increased between the 1930s and 1990s, leading to a 14% overall increase in habitat. In general, the models for the two time periods are fairly similar. Cliffs for nesting appear to be limited in the landscape: I identified only 129 ha of cliffs concentrated in the northeast and southwest portions of the study area. However, my method likely underestimated the amount of available cliff habitat. Similarly, I believe my estimate of foraging habitat is conservative given the small wetlands widely distributed throughout the study area.

Marten. I assumed the limiting factor for marten was winter habitat, and further that winter habitat would receive the most use on a yearly basis for foraging and denning, although younger stands and open areas may receive some foraging use in summer. I selected for mature/overmature mixed conifer, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir stands $\geq$15 ha. Ability to select habitat based on stand density would probably have improved this model. Predicted habitat declined by 14% between the 1930s and 1990s, although the decline is much sharper if the amount of predicted habitat at 2 ha MMU is examined. Clearly, the minimum size criterion eliminated many stands in the 2 ha MMU layer, highlighting the fact that examination of mature/overmature forests at a finer resolution will reveal more extreme patterns of fragmentation in the Seeley-Swan. Nonetheless, a comparison of spatial statistics for marten habitat at 16 ha MMU reveals fairly heavy fragmentation in the past 60 years. Whereas in the 1930s, the largest block of predicted habitat was in the north end of the Swan Valley, today the Clearwater Divide area has large, contiguous blocks of habitat.

This illustrates the conversion of mature/overmature forests in the northern Swan Valley to younger stands, while young stands of lodgepole pine in the Clearwater Divide region have matured and become suitable habitat.

<u>Fisher.</u> In modeling fisher habitat, I assumed that mature/overmature mixed conifer, Douglas-fir, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir stands would be used most heavily on a yearly basis, while pole stands of the same forest types would be used in winter. From the above set, I selected only the areas within 400 m of perennial streams, lakes, and marshes (Heinemeyer 1993). A 10% decline in predicted habitat between the 1930s and 1990s was observed; although the amount of winter habitat increased by 16%, year-round habitat decreased by 22%. Overall, predicted fisher habitat became increasingly fragmented.

<u>Wolverine.</u> For the wolverine, I assumed that food availability was the limiting factor in habitat use; Hornocker and Hash (1981) believed that food availability was the primary factor determining the movements and range of wolverine in the South Fork of the Flathead. In modeling habitat, I selected pole and mature/overmature mixed conifer, Douglas-fir, lodgepole pine, and Engelmann spruce/subalpine fir stands as well as all whitebark pine/Engelmann spruce/ subalpine fir stands. I selected only those stands ≥1300 m on easterly and southerly aspects (45-225°). Finally, I identified ecotonal areas -- barren, grass, shrub, and rocky woodland types within 60 m of selected forest stands. Somewhat surprisingly, I predicted a 25% decrease in the amount of habitat between the 1930s and 1990s. This is probably due to the distribution of cover types in the two time periods: In the 1930s, the high elevations were almost uniformly classified as subalpine or barren, while cover types in the 1990s were more diverse, including a fairly large area classified as seedling/sapling or grass. Although wolverines have been seen in riparian areas and pastures of the Swan Valley bottom (USDA:FS 1994a), I felt that an elevation cutoff was appropriate based on Hornocker and Hash's (1981) findings that wolverines used higher elevations (mean 1371 m in

winter, when the lowest elevations were used). However, these observations at lower elevations can be interpreted as an indication that barriers to travel are not a significant factor in the Seeley-Swan.

Lynx. I included denning and foraging habitat in the lynx model, defining denning habitat as mature/overmature Engelmann spruce/subalpine fir or lodgepole pine stands on northeasterly aspects ($\geq$50% of stand 0-135° or 315-360°) and foraging habitat as pole-sized lodgepole pine. The model would have been improved if I had been able to select dense lodgepole sapling stands as foraging habitat, as these stands would be higher quality snowshoe hare habitat (Koehler 1990). Overall, predicted lynx habitat declined by 36% over the past 60 years; this decline was sharper for foraging (49%) than denning (21%) habitat. However, I undoubtedly underestimated the amount of foraging habitat by failing to include dense lodgepole sapling stands in the Seeley-Swan. Although lynx habitat appears to have become more fragmented over time, the largest patch index was higher for the 1990s than the 1930s, reflecting a large patch of denning habitat in the Clearwater Divide area. In both time periods, denning habitat was more plentiful on the east side of the Mission Mountains, and foraging habitat was more common in the valley bottoms. Note that for the lynx, wolverine, fisher, and marten, road densities could be an important consideration in habitat modeling because higher densities increase trapping vulnerability.

Gray Wolf. For the wolf, I assumed that the limiting factors were prey availability and vulnerability to human-caused mortality. I modeled habitat by selecting polygons within big game winter ranges and with total road density $\leq$3 mi/mi$^2$ (Pletscher, pers.comm.). Lacking a complete layer of roads for the 1930s, I assumed that all areas had road densities $\leq$3 mi/mi$^2$ at that time. However, it could also be assumed that much of the predicted habitat for the 1930s was unsuitable because the majority of human settlements fell within this habitat and complete predator control was the norm during that period. Thus, wolves might have been more actively pursued by humans at that time. As modeled, predicted habitat declined sharply (91%) between the 1930s and 1990s. This is also perhaps the most striking case of fragmentation; few secure areas remain for wolves in the Seeley-Swan landscape based on these criteria.

Grizzly Bear. This model is based almost entirely on security: I assumed that the entire study area (except water, agricultural, and urban areas) was potentially suitable habitat for the grizzly bear, which uses a diversity of habitats (USDI:FWS 1993). I then eliminated areas with total road density $>2$ mi/mi$^2$ (Mace and Manley 1993). Again, for the 1930s I assumed that no areas had road densities above the cutoff value; thus, the amount of predicted habitat showed a 52% decline over time. In the 1990s, secure areas for grizzlies occur mostly at the higher elevations. This highlights potential for human-bear conflicts (as already confirmed by past experiences in the Seeley-Swan) because some of the more productive habitats for grizzly bears, including riparian areas, occupy the valley bottoms. Note that security areas may be defined in a number of ways; I also mapped security areas as those areas $>500$ m from any road, and found that selected areas corresponded well to areas with total road density $\leq 2$ mi/mi$^2$.

Mountain Goat. I selected the following cover types above 1845 m: barren, rocky woodland, grass, shrub, and whitebark pine/Engelmann spruce/subalpine fir (no elevation limit was placed on the latter). Predicted habitat decreased by 22% between the 1930s and 1990s, probably for reasons similar to those outlined for the wolverine. Spatial distribution of habitat is very similar for the two time periods.

Shiras Moose. I created a 150 m buffer around all streams, lakes, and marshes in the Seeley-Swan, then selected for seedling/sapling and mature/overmature forest types, as well as shrub and broadleaf types, within the buffer. Areas with mean slope $>50\%$ were eliminated. Predicted habitat increased slightly (3%) between the 1930s and 1990s. Most probably, area occupied by seedling/sapling stands increased while area in mature/overmature forest decreased, and thus the amount of predicted habitat was held relatively constant. In both periods, habitat was well distributed throughout the study area.

**Summary of Habitat Trends**

Fragmentation of habitat is the major threat to most species in the temperate zone; both of its two components, overall loss and insularization of habitat, cause extinctions (Wilcove et al. 1986). Fragmentation may affect species richness within a landscape, population trends for individual species, and biological diversity overall (Morrison et al. 1992). By now, the unfortunate reader has been bludgeoned with the message of increasing habitat fragmentation in the Seeley-Swan landscape, as predicted by 20 simple wildlife models. For most species, habitat was less contiguous, occurred in smaller patches, and occupied less area in the 1990s than in the 1930s. Although mean nearest-neighbor distances tended to be smaller in the 1990s, suggesting that patches were less isolated, the increase in patch number balanced this out: It appears that large patches were dissected into numerous closely-spaced patches. In the 1930s, patches were farther apart, but they were much larger. It has already been noted that mean patch size for vegetation polygons in the 1930s greatly exceeded that for the 1990s despite an equivalent MMU (mean 153 ha versus 72 ha). Several factors may contribute to this difference: 1) single-pixel water polygons preserved in the merging process for the 1990s may be bringing the average down; 2) the mappers in the 1930s may not have adhered to their designated MMU; and 3) most probably, stands were naturally larger before management activities became a dominant force in shaping landscape patterns. Although I suspect mappers tended to generalize stand boundaries in the 1930s, I believe this difference in stand size is more than just an artifact of mapping procedures, and instead represents a significant change in the Seeley-Swan landscape over time.

Most of the predicted declines in habitat stem from two factors; decreasing area occupied by mature/overmature forests is responsible for the majority of predicted declines, whereas increasing road density led to decreases in secure habitat for grizzly bears and wolves. The importance of mature/overmature forest was guaranteed when I selected species based primarily on management status -- a number of species are of special concern because they are known or suspected to be tied to older forests. Road density and other factors related to human disturbance could have been incorporated into several more models, and would have resulted in a slightly more grim picture of habitat conditions in the Seeley-Swan for these species.

**Scale**

As MMU increased, the amount of predicted habitat for the pileated woodpecker steadily declined; the net difference in predicted habitat between 2 ha MMU and 400 ha MMU was 6519 ha. Figure 4-25 shows changes in the spatial arrangement of predicted habitat with increasing MMU. As a rule, the amount of predicted habitat can be expected to decline with increasing MMU, as was seen for the pileated woodpecker. In addition, patches of habitat may appear to be more isolated as the smaller patches between larger concentrations are eliminated. Thus, the importance of selecting a map resolution appropriate to the size of the landscape and the purpose of the analysis is emphasized, as it was in Chapter 3.

Stoms (1992) also examined the effects of increasing MMU on assessments of biodiversity, finding that as a habitat map was generalized, the number of habitat types tended to decrease, as did the number of species predicted to occur within grid cells. Stoms felt that the change in spatial distribution of species richness predictions was a more serious issue than changes in amount of predicted habitat, because such maps may be incorporated in selection of nature reserves. His concerns illustrate how a map's future utility hinges on its construction; although selection of an appropriate MMU is not the only methodological issue, it is undoubtedly an important consideration.

Selection of mapping resolution, as well as study area extent, is also important in terms of how individual species perceive the landscape. Animals may identify and use patches and resources at varying scales depending on factors like body size, resource orientation, and home range size (Morrison et al. 1992). For wide-ranging species like the grizzly bear and gray wolf, the Seeley-Swan landscape alone provides insufficient area to support viable populations; thus, landscape context and connectivity at a regional scale become important considerations. Other species like the harlequin duck may key into microhabitat features that are difficult to capture in a GIS database; models for such species may not adequately represent habitat conditions. Ideally, habitat evaluations should be conducted at a variety of scales; there is no single inherent scale at which ecological phenomena should be examined (Levin 1992).

### Species Richness

The number of species predicted to be present was higher for a larger area of the Seeley-Swan landscape in the 1930s than in the 1990s. Species richness values tended to be higher in the 1930s than in the 1990s largely because of the more widespread occurrence of mature/overmature forest 60 years ago. Species counts were generally highest at lower elevations in the 1930s (Fig. 4-26) and at middle elevations in the 1990s (Fig. 4-27). Historically, the valley bottom had the highest species concentrations, with large expanses of habitat in the northern end of the study area predicted to contain 10 or more species. Today, concentrations can be seen along the slopes of the Mission and Swan Ranges, as well as the Clearwater Divide. Those patches predicted to currently contain habitat for many species in the valley bottom should be examined closely for inclusion in the network of protected areas.

Simple species counts may not be sufficient for management purposes where some species are assigned higher priorities than others. In such cases, species counts may be weighted. As an example, values may be assigned to species based on management status, with endangered species weighted the heaviest and species without special designation accorded the least weight. Similar schemes may be devised for various applications, and may highlight areas not assigned high values according to raw counts.

**Management Applications**

As with any model, the outputs of this exercise are only as good as the inputs; in this case, the latter include my interpretation of existing literature and the attributes of the GIS database at my disposal. Availability of literature on habitat selection is not the only factor influencing model quality; accuracy of habitat maps is influenced by interactions between MMU, resolution of source data, map generalization, and analyst skill, among other factors (Lodwick et al. 1990). Thus, these maps represent my own interpretations and synthesis of existing information; it remains to the reader to determine the validity of my approach and conclusions.

Although habitat models may not offer new information, they are a means of organizing our collective knowledge and arriving at first approximations of habitat conditions within a given area (Scott et al. 1991). The utility of GIS habitat modeling to biologists is twofold. First, GIS modeling focuses biologists on the assumptions they make about wildlife-habitat relationships, then maps those assumptions over a broad scale, thus serving as a tool for visualization. A variety of maps based on different assumptions can be produced rapidly and efficiently, allowing exploratory analyses that would be cumbersome using traditional techniques. Further, through GIS modeling, a base map is produced which can then be used as a focal point for future field surveys, and then for model validation and improvement. The power of a GIS lies in its ability to analyze large areas quickly and consistently once the necessary databases have been constructed; such broad-scale analyses are among the most urgent tasks facing conservation biologists today.

Table 4-1. Twenty wildlife species for which habitat was modeled using a GIS. Species were selected to represent a variety of taxa, ecological roles, and habitats used. The U.S. Forest Service Region 1 sensitive species list was emphasized (Jolly 1994), and several management indicator species for the Flathead National Forest were also included (USFS 1985, 1992).

| | | STATUS[a] | | |
|---|---|---|---|---|
| COMMON NAME | SCIENTIFIC NAME | USFWS[b] | USFS R1 | MT |
| Tailed Frog | *Ascaphus truei* | | | NG |
| Harlequin Duck | *Histrionicus histrionicus* | C2 | S | MB |
| Common Loon | *Gavia immer* | | S | P |
| Townsend's Warbler | *Dendroica townsendi* | | | P |
| Black-backed Woodpecker | *Picoides arcticus* | | S | P |
| Pileated Woodpecker | *Dryocopus pileatus* | | MIS | P |
| Flammulated Owl | *Otus flammeolus* | | S | P |
| Boreal Owl | *Aegolius funereus* | | S | P |
| Barred Owl | *Strix varia* | | MIS | P |
| Northern Goshawk | *Accipiter gentilis* | C2 | | P |
| Bald Eagle | *Haliaeetus leucocephalus* | LT | E | P |
| Peregrine Falcon | *Falco peregrinus anatum* | LE | E | E |
| Marten | *Martes americana* | | MIS | FB |
| Fisher | *Martes pennanti* | | S | FBRH |
| Wolverine | *Gulo gulo* | C2 | S | FBRH |
| Lynx | *Felis lynx* | C2 | S | FBRH |
| Gray Wolf | *Canis lupus* | LE | E | E |
| Grizzly Bear | *Ursus arctos* | LT | T | GARH |
| Mountain Goat | *Oreamnos americanus* | | | GA |
| Shiras Moose | *Alces alces shirasi* | | | GA |

[a] Status taken from Jolly (1994) and Montana Natural Heritage Database, Vertebrate Character Abstracts (December 1994).
[b] **USFWS**: LE = listed endangered; LT = listed threatened; C2 = Category 2.
**USFS R1**: E = endangered; T = threatened; S = sensitive; MIS = management indicator species, Flathead NF. **MT**: E = endangered; P = protected; FB = furbearer; FBRH = furbearer, restricted harvest; GA = game animal; GARH = game animal, restricted harvest; NG = nongame species; MB = migratory bird.

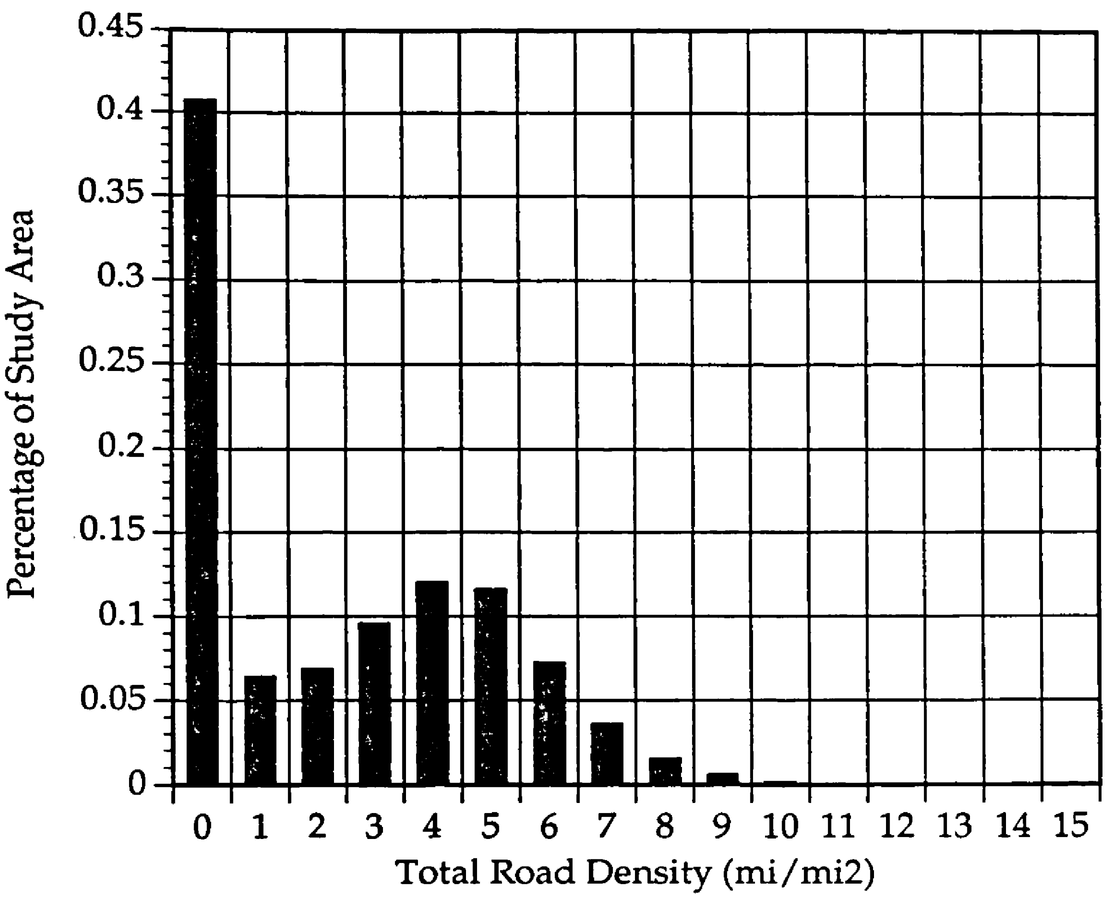


Figure 4-1. Frequency distribution for road density in the Seeley-Swan landscape, northwestern Montana, 1994. Road densities were calculated on a pixel-by-pixel basis using a moving-circle technique (Appendix B).

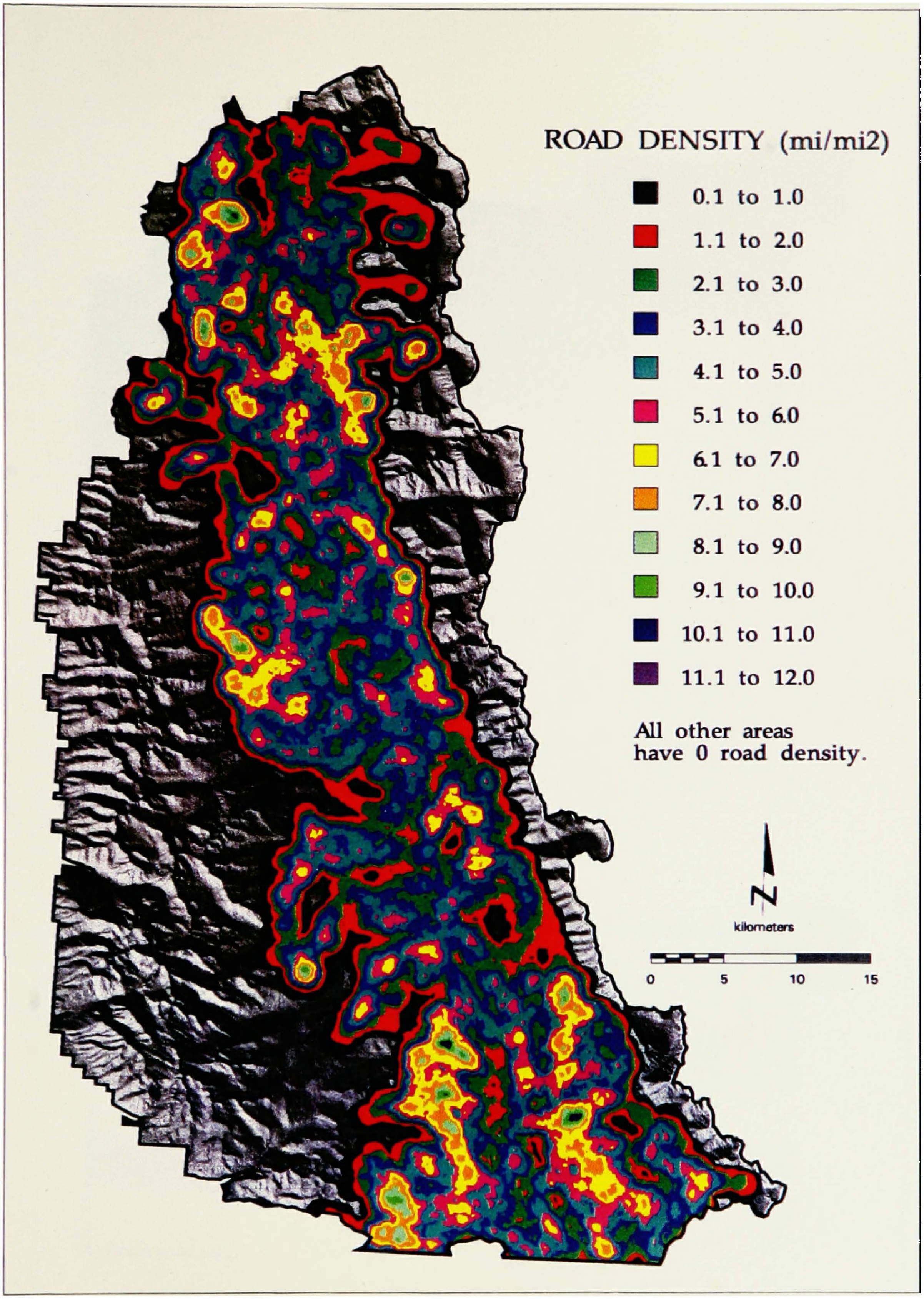


Figure 4-2. Road densities in 1994 for the Seeley-Swan landscape, northwestern Montana, calculated using a moving-circle technique.

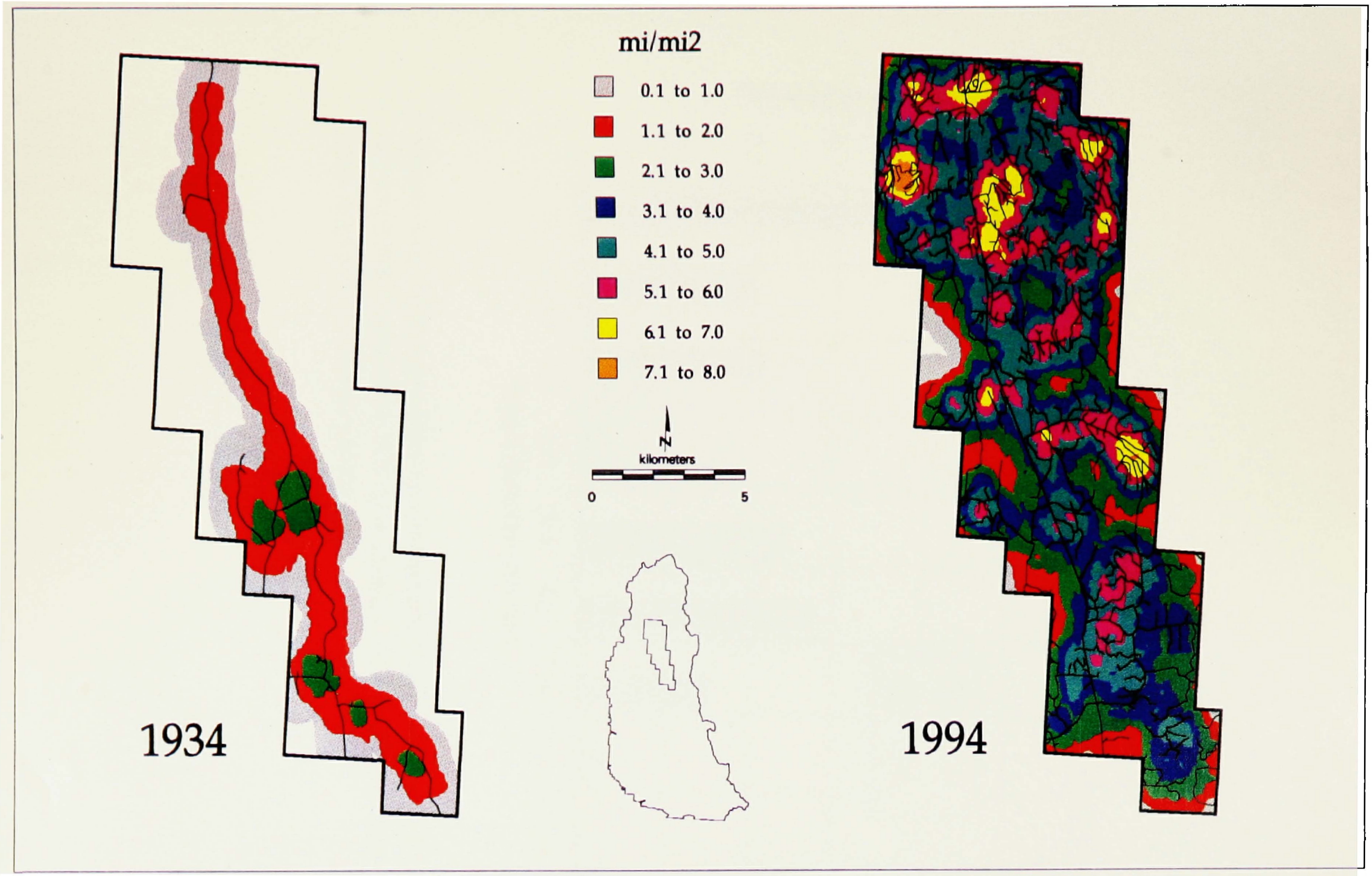


Figure 4-3. Increase in total road density over time in the central Swan Valley, northwestern Montana.

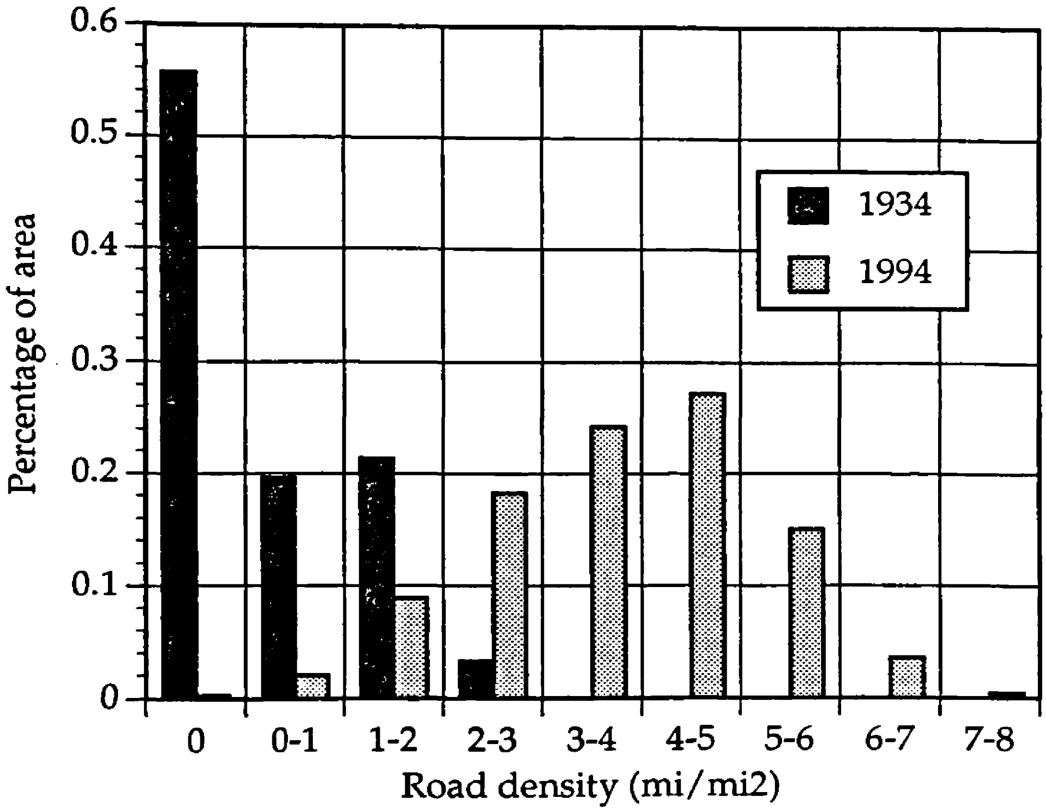


Figure 4-4. Road densities in the central Swan Valley, 1934 versus 1994, based on a 570 m moving circle analysis. The total analysis area was 14,305 ha.

Table 4-2. Sensitivity analysis for lynx denning habitat, illustrating influence of modeling rules on amount of predicted habitat.

| | | | SELECTED HABITAT[a] | | | |
|---|---|---|---|---|---|---|
| | | | Number of Patches | | Hectares | |
| PREMISE[b] | VARIABLES | GIS DEFINITION OF HABITAT | 1930s | 1990s | 1930s | 1990s |
| 1. Lynx den in mature/old-growth | vegetation | • all m/om stands | 457 | 5537 | 94,199 | 72,503 |
| PICO[c] or PIEN/ABLA stands. | | • all m/om PICO and PIEN/ABLA stands | 117 | 2121 | 24,664 | 34,624 |
| 2. Lynx use habitats >1463 m. | vegetation, | • m/om stands ≥1460 m | 199 | 3128 | 44,047 | 37,879 |
| | elevation | • m/om PICO, PIEN/ABLA ≥1460 m | 82 | 1631 | 22,174 | 26,049 |
| 3. Lynx prefer to den on north | vegetation, | • m/om - mean aspect[d] | 119 | 2241 | 21,719 | 27,552 |
| and northeast aspects. | aspect | • m/om - majority aspect | 225 | 2513 | 48,794 | 36,088 |
| | | • m/om PICO, PIEN/ABLA - mean aspect | 36 | 798 | 6295 | 10,453 |
| | | • m/om PICO, PIEN/ABLA - majority aspect[e] | 88 | 941 | 21,127 | 17,071 |
| | vegetation, | • m/om ≥1460 m - mean aspect | 53 | 1052 | 9240 | 12,740 |
| | aspect, and | • m/om ≥1460 m - majority aspect | 101 | 1357 | 26,641 | 22,104 |
| | elevation | • m/om PICO, PIEN/ABLA ≥1460 m - mean aspect | 26 | 555 | 5639 | 6269 |
| | | • m/om PICO, PIEN/ABLA ≥1460 m - majority aspect | 64 | 703 | 19,156 | 13,431 |
| 4. Denning sites should be near | vegetation, | • ≤1 km from denning to foraging habitat | 55 | 675 | 13,946 | 15,153 |
| foraging habitat (pole-sized PICO). | aspect, and | • ≤500 m from denning to foraging habitat | 44 | 507 | 11,942 | 13,560 |
| | distance | • ≤100 m from denning to foraging habitat | 34 | 263 | 10,594 | 10,497 |

[a] Out of a potential 247,925 ha and 1622 vegetative patches for the 1930s (16 ha); same area but 24,903 vegetative patches for the 1990s (2 ha).
[b] Drawn from Koehler (1990) and Koehler and Brittell (1990).
[c] PICO = lodgepole pine, PIEN = Engelmann spruce, ABLA = subalpine fir, m/om = mature/overmature.
[d] Aspects ≤135° or ≥315° selected; see text (northern goshawk) for discussion of mean versus majority aspect calculations.
[e] Used in lynx model (Appendix C) as well as calculation of distance between denning and foraging habitat.

Table 4-3. Amount of habitat predicted to occur for 20 species in the Seeley-Swan landscape, 1930s versus 1990s. For the 1990s, models were constructed at two minimum mapping units (MMUs) to demonstrate the effect of spatial resolution on habitat predictions. The 1930s maps were built at a 16 ha MMU.

| | PREDICTED HABITAT (ha) | | | |
|---|---|---|---|---|
| | | 1990s | | |
| SPECIES | 1930s | 2 ha MMU | 16 ha MMU | % Change[a] |
| Tailed Frog | 3891 | 2553 | 2566 | -34 |
| Harlequin Duck | 615 | 271 | 271 | -56 |
| Common Loon | 1768 | 1842 | 1777 | +1 |
| Townsend's Warbler | 80,141 | 55,636 | 55,187 | -31 |
| Black-backed Woodpecker | 98,327 | 68,689 | 70,217 | -29 |
| Recent burn | 4439 | 193 | 149 | -97 |
| Other | 93,888 | 68,496 | 70,068 | -25 |
| Pileated Woodpecker | 94,136 | 57,445 | 56,873 | -40 |
| Flammulated Owl | 13,995 | 1767 | 1686 | -88 |
| Boreal Owl | 53,241 | 42,835 | 44,566 | -16 |
| Barred Owl | 83,220 | 53,873 | 57,906 | -30 |
| Northern Goshawk | 2800 | 8312 | 4043 | +44 |
| Bald Eagle | 32,484 | 7379 | 8454 | -74 |
| Peregrine Falcon | 5030 | 7571 | 5729 | +14 |
| Nesting | 129 | 129 | 129 | 0 |
| Foraging | 4900 | 7442 | 5600 | +14 |
| Marten | 77,915 | 45,536 | 66,854 | -14 |
| Fisher | 66,693 | 61,294 | 59,960 | -10 |
| Winter | 20,601 | 25,696 | 23,856 | +16 |
| Year-round | 46,092 | 35,598 | 36,104 | -22 |
| Wolverine | 116,548 | 78,844 | 87,462 | -25 |
| Lynx | 44,445 | 28,741 | 28,614 | -36 |
| Denning | 21,127 | 17,071 | 16,679 | -21 |
| Foraging | 23,318 | 11,670 | 11,935 | -49 |
| Gray Wolf | 46,086 | 7013 | 4335 | -91 |
| Grizzly Bear | 243,463 | 122,781 | 117,127 | -52 |
| Mountain Goat | 63,962 | 47,173 | 49,677 | -22 |
| Shiras Moose | 51,146 | 55,603 | 52,540 | +3 |

[a] (1930s ha - 1990s ha (16 ha MMU) / 1930s ha) * 100

Table 4-4. Spatial statistics for wildlife habitat in the Seeley-Swan landscape, 1930s and 1990s (minimum mapping unit 16 ha).

| | Number Patches | | Largest Patch (%) | | Mean Patch Size (SD) -- ha | | | | Mean Nearest Neighbor (SD) -- m | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| SPECIES | 1930s | 1990s | 1930s | 1990s | 1930s | | 1990s | | 1930s | | 1990s | |
| Tailed frog | 514 | 1003 | 4.7 | 2.7 | 8 | (15) | 3 | (5) | 317 | (505) | 199 | (304) |
| Harlequin duck | 120 | 197 | 13.5 | 8.2 | 5 | (11) | 1 | (3) | 1275 | (1989) | 671 | (1617) |
| Common loon | 55 | 56 | 22.1 | 21.4 | 32 | (73) | 32 | (70) | 1979 | (2125) | 1664 | (2048) |
| Townsend's warbler | 108 | 376 | 19.1 | 5.0 | 742 | (2147) | 147 | (321) | 488 | (795) | 238 | (318) |
| Black-backed woodpecker | 130 | 344 | 38.5 | 13.5 | 756 | (3517) | 204 | (804) | 1150 | (2376) | 355 | (2398) |
| Recent burn | 28 | 3 | 1.4 | 0.1 | 159 | (286) | 50 | (27) | 3947 | (3906) | 18,409 | (17,961) |
| Other | 102 | 341 | 38.5 | 13.5 | 920 | (3952) | 205 | (808) | 382 | (518) | 196 | (261) |
| Pileated woodpecker | 98 | 389 | 42.1 | 4.8 | 961 | (4199) | 146 | (316) | 372 | (524) | 210 | (289) |
| Flammulated owl | 53 | 29 | 35.7 | 11.2 | 264 | (703) | 58 | (45) | 1085 | (2041) | 1518 | (2241) |
| Boreal owl | 88 | 274 | 12.0 | 6.1 | 605 | (1218) | 163 | (325) | 585 | (996) | 234 | (330) |
| Barred owl | 89 | 290 | 44.5 | 14.3 | 935 | (4084) | 200 | (675) | 406 | (526) | 186 | (226) |
| Northern goshawk | 21 | 58 | 13.5 | 16.1 | 133 | (118) | 70 | (120) | 4263 | (4073) | 1847 | (1820) |
| Bald eagle | 28 | 206 | 67.3 | 7.6 | 1160 | (4085) | 41 | (102) | 416 | (382) | 215 | (290) |
| Peregrine falcon | 317 | 813 | 15.0 | 5.8 | 16 | (55) | 7 | (20) | 679 | (1209) | 381 | (823) |
| Nesting | 29 | 29 | 0.4 | 0.4 | 5 | (5) | 5 | (5) | 1897 | (3152) | 1897 | (3152) |
| Foraging | 288 | 784 | 15.0 | 5.8 | 17 | (57) | 7 | (20) | 556 | (666) | 325 | (496) |
| Marten | 93 | 336 | 19.6 | 13.5 | 838 | (2275) | 199 | (704) | 446 | (732) | 236 | (289) |
| Fisher | 567 | 1508 | 10.5 | 5.3 | 118 | (458) | 40 | (135) | 333 | (661) | 200 | (318) |
| Winter | 306 | 705 | 2.5 | 2.3 | 67 | (178) | 34 | (95) | 347 | (695) | 230 | (373) |
| Year-round | 261 | 803 | 10.5 | 5.3 | 177 | (642) | 45 | (162) | 314 | (616) | 171 | (255) |
| Wolverine | 41 | 232 | 64.5 | 28.9 | 2843 | (11,854) | 377 | (2196) | 225 | (351) | 133 | (246) |
| Lynx | 150 | 249 | 7.8 | 18.4 | 296 | (483) | 115 | (386) | 868 | (1884) | 989 | (1417) |
| Denning | 53 | 123 | 7.8 | 18.4 | 399 | (646) | 136 | (489) | 993 | (2341) | 830 | (1292) |
| Foraging | 97 | 126 | 4.1 | 6.5 | 240 | (353) | 95 | (245) | 800 | (1576) | (1144) | (1512) |
| Gray wolf | 12 | 59 | 79.3 | 27.3 | 3841 | (10,110) | 73 | (165) | 273 | (209) | 694 | (1051) |
| Grizzly bear | 10 | 35 | >99.9 | 65.6 | 24,346 | (73,015) | 3346 | (14,060) | 41 | (11) | 380 | (937) |
| Mountain goat | 27 | 60 | 62.9 | 35.1 | 2369 | (8449) | 828 | (2944) | 853 | (1918) | 227 | (392) |
| Shiras moose | 501 | 1277 | 12.6 | 16.0 | 102 | (500) | 41 | (268) | 169 | (236) | 110 | (164 |

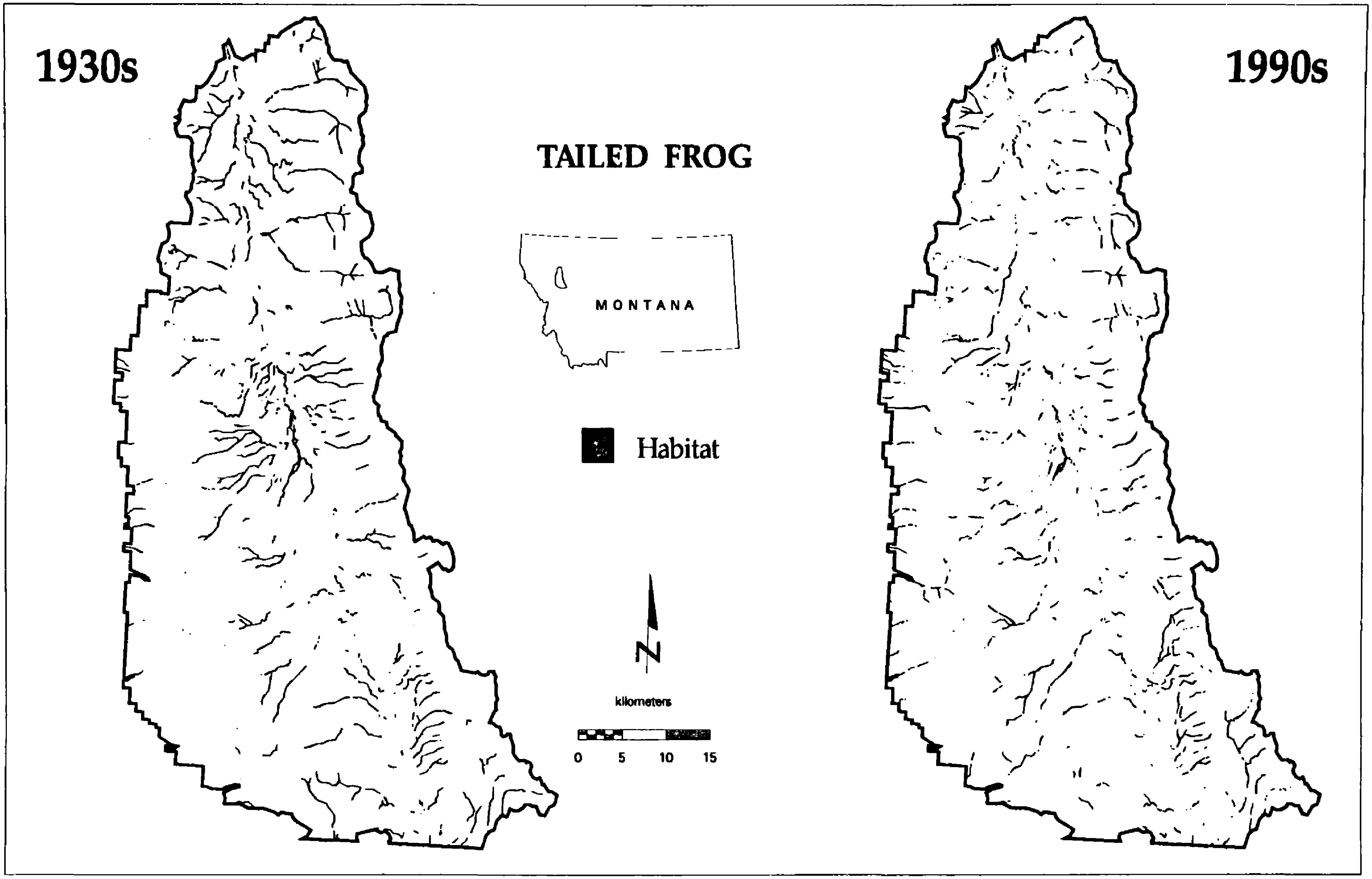


Figure 4-5. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the tailed frog (*Ascaphus truei*).

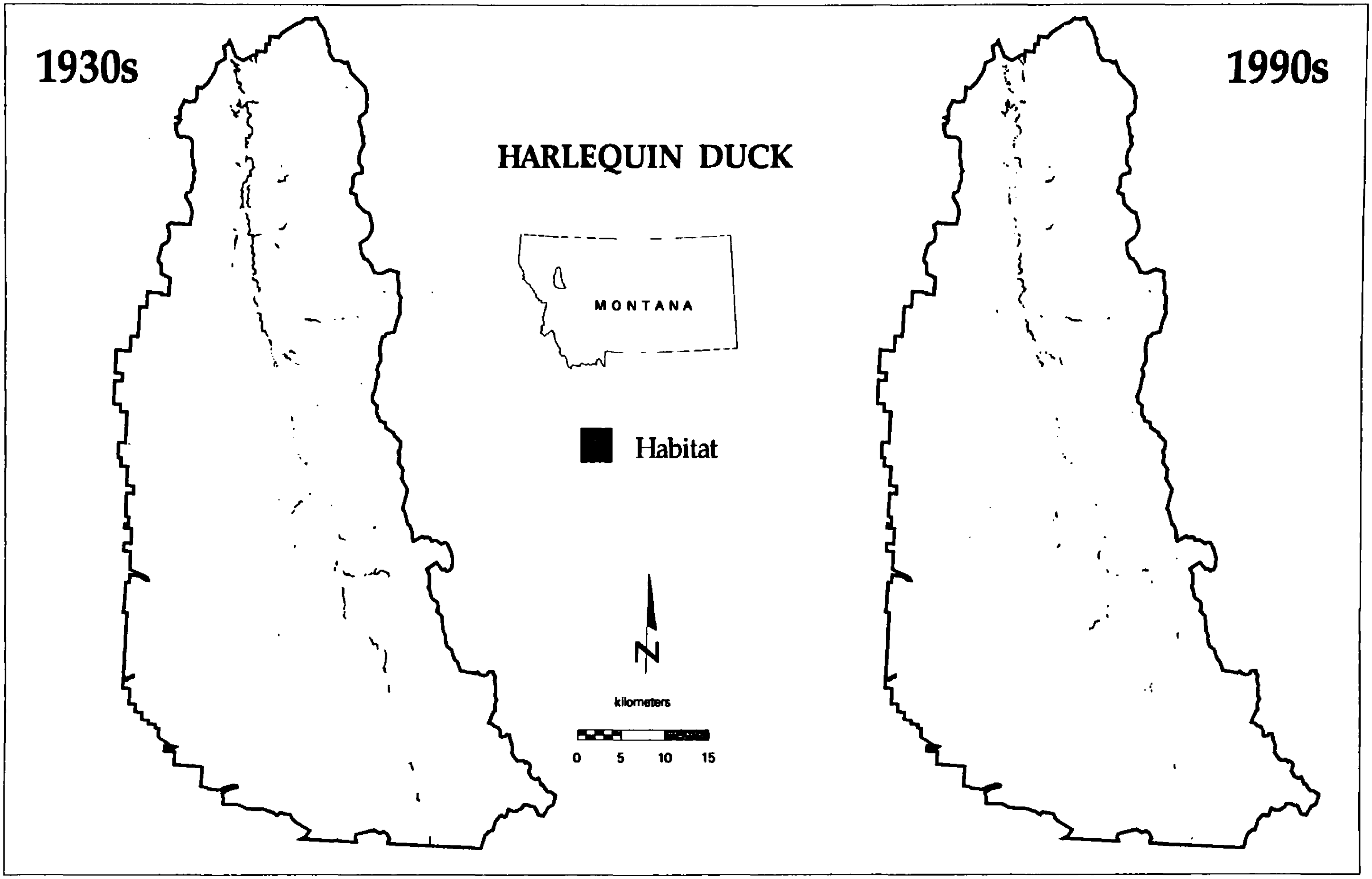


Figure 4-6. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the harlequin duck (*Histrionicus histrionicus*).

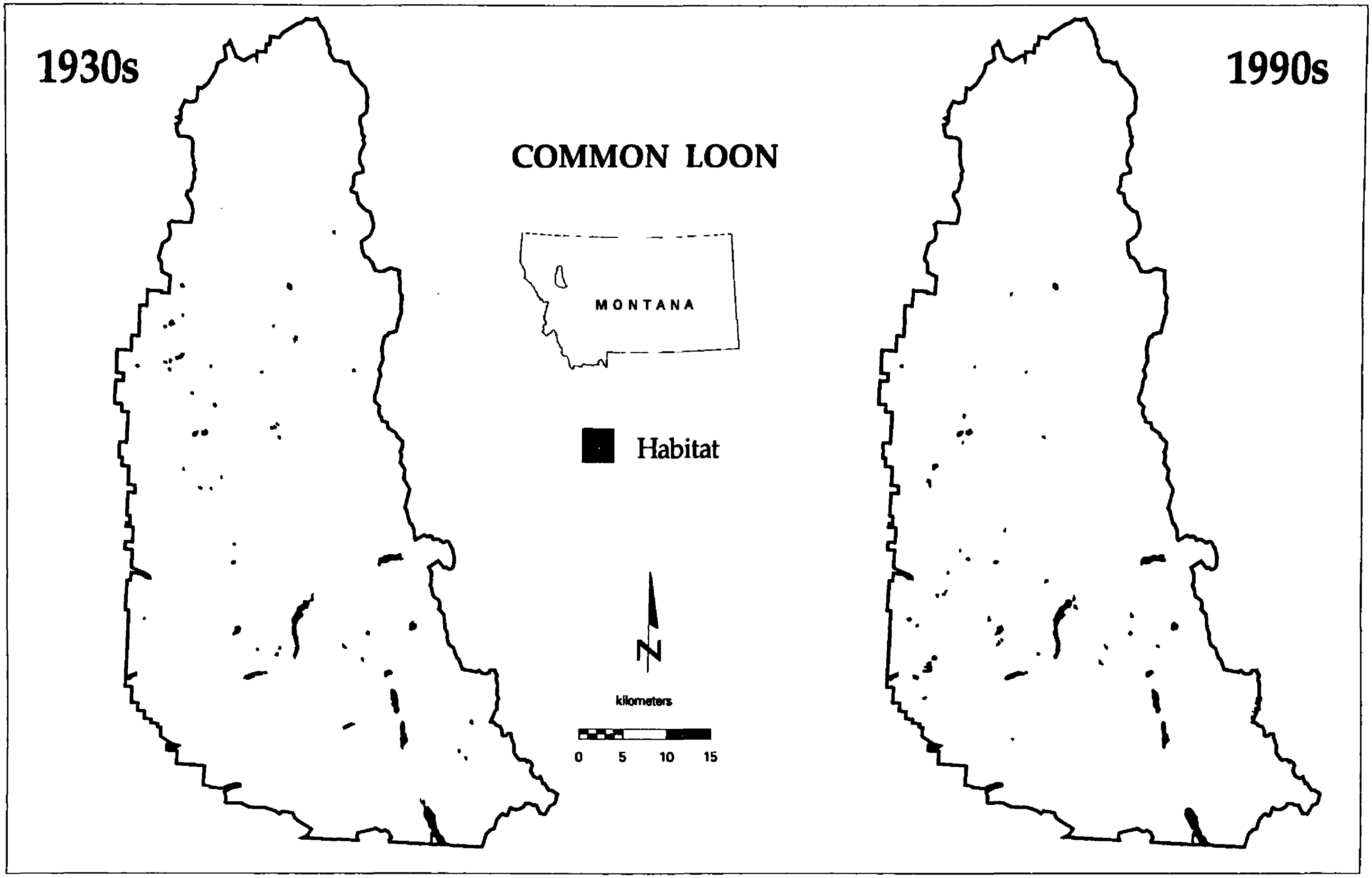


Figure 4-7. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the common loon (*Gavia immer*).

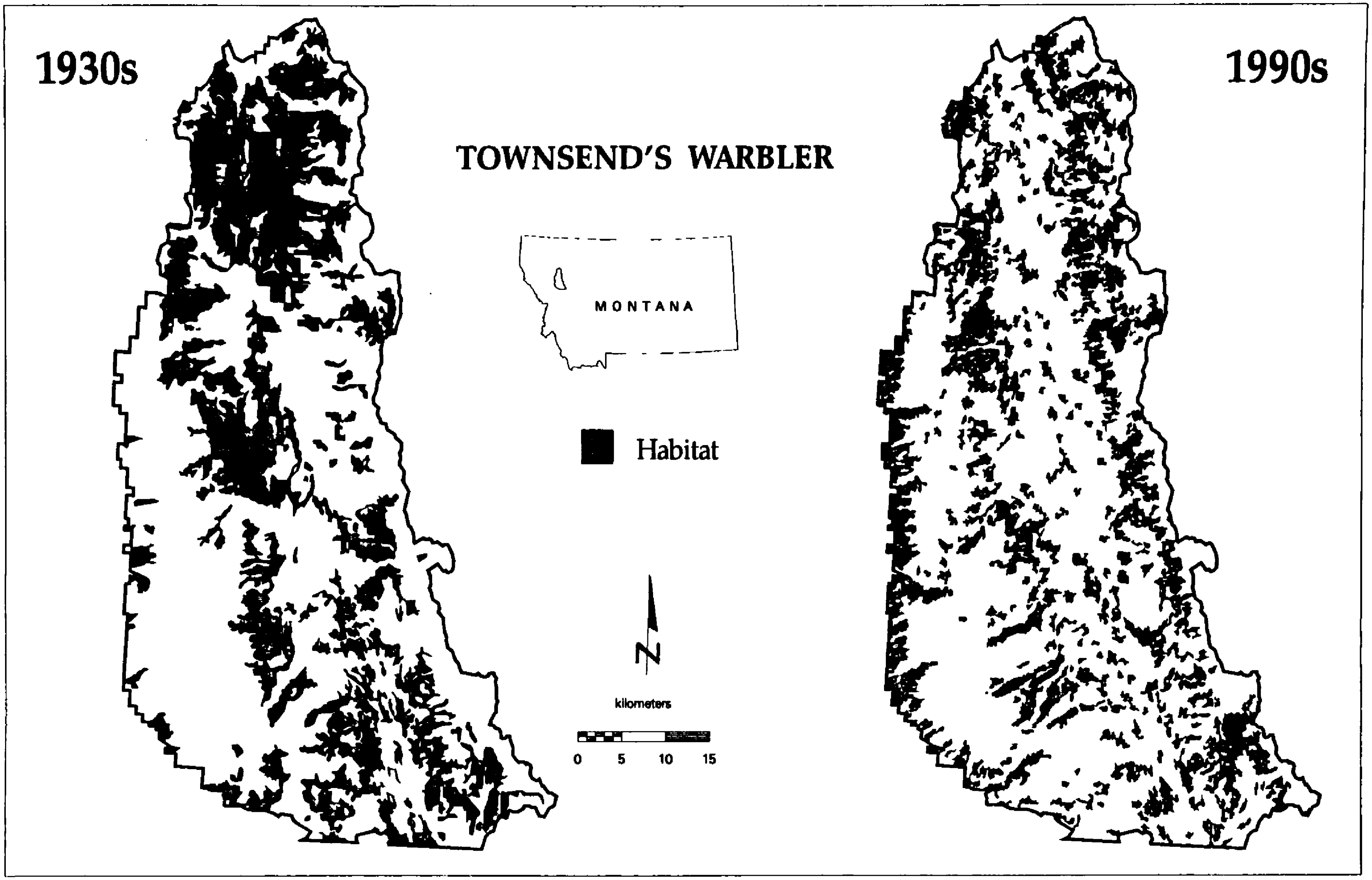


Figure 4-8. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the Townsend's warbler (*Dendroica townsendi*).

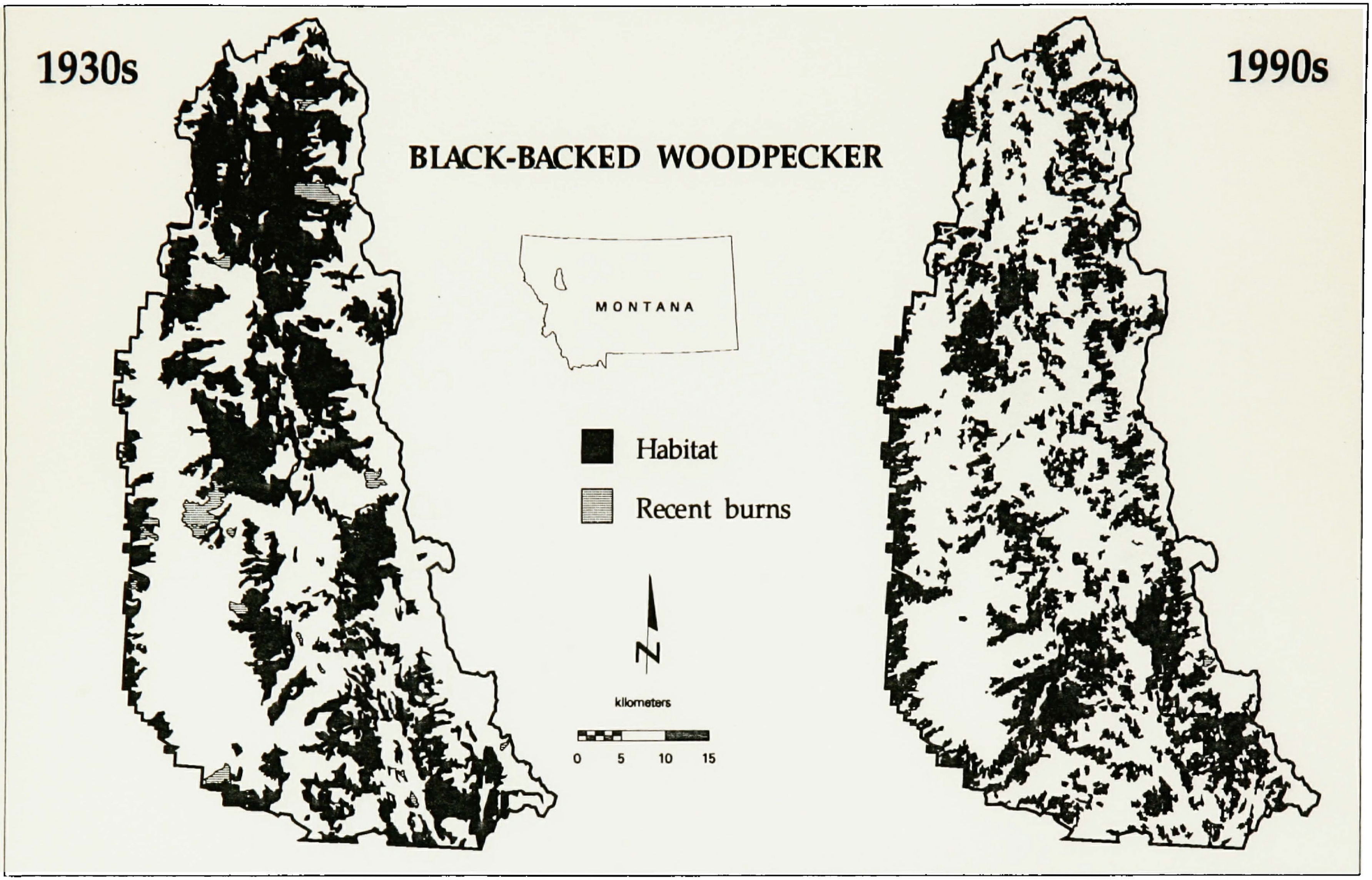


Figure 4-9. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the black-backed woodpecker (*Picoides arcticus*).

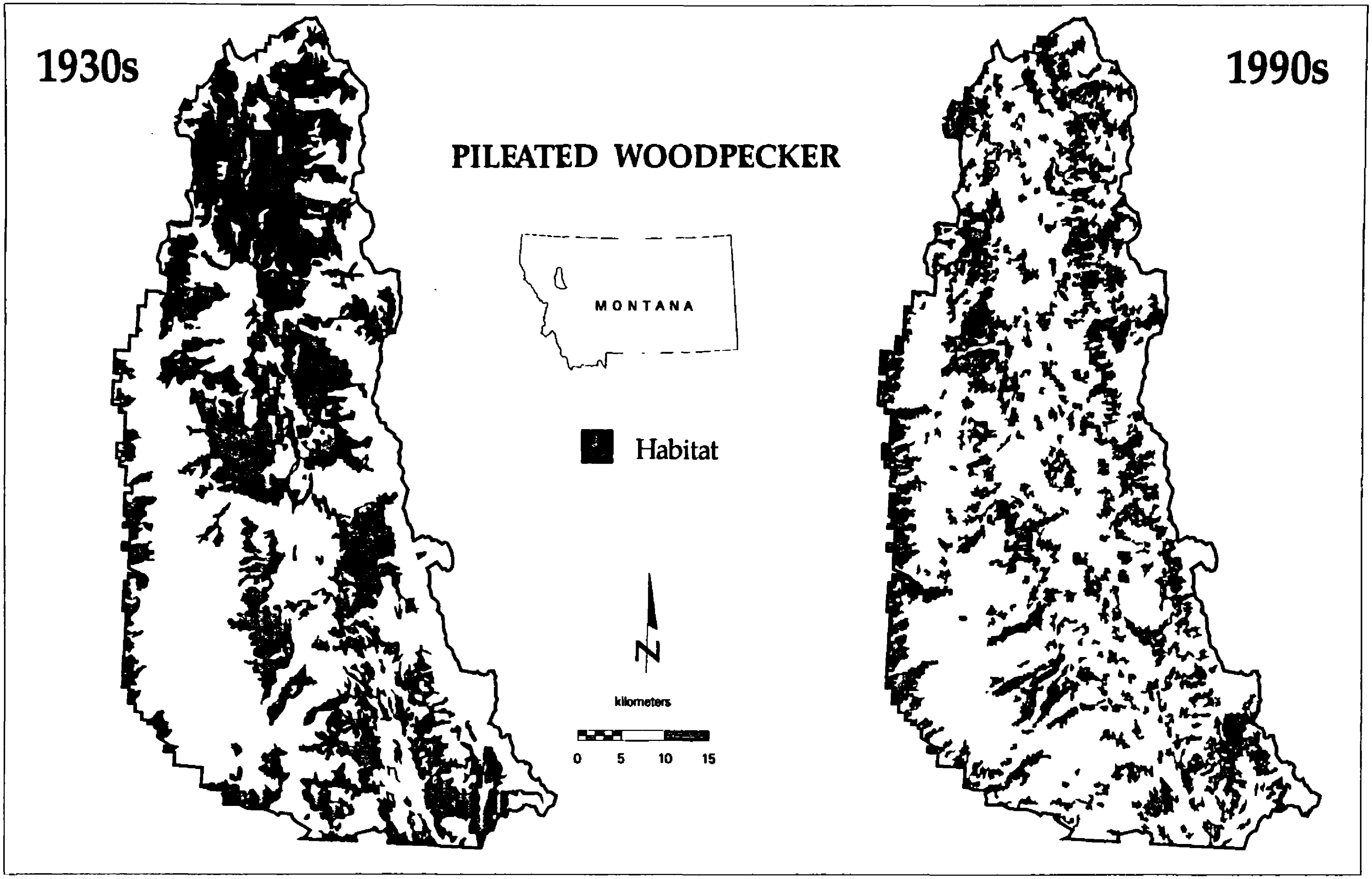


Figure 4-10. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the pileated woodpecker (*Dryocopus pileatus*).

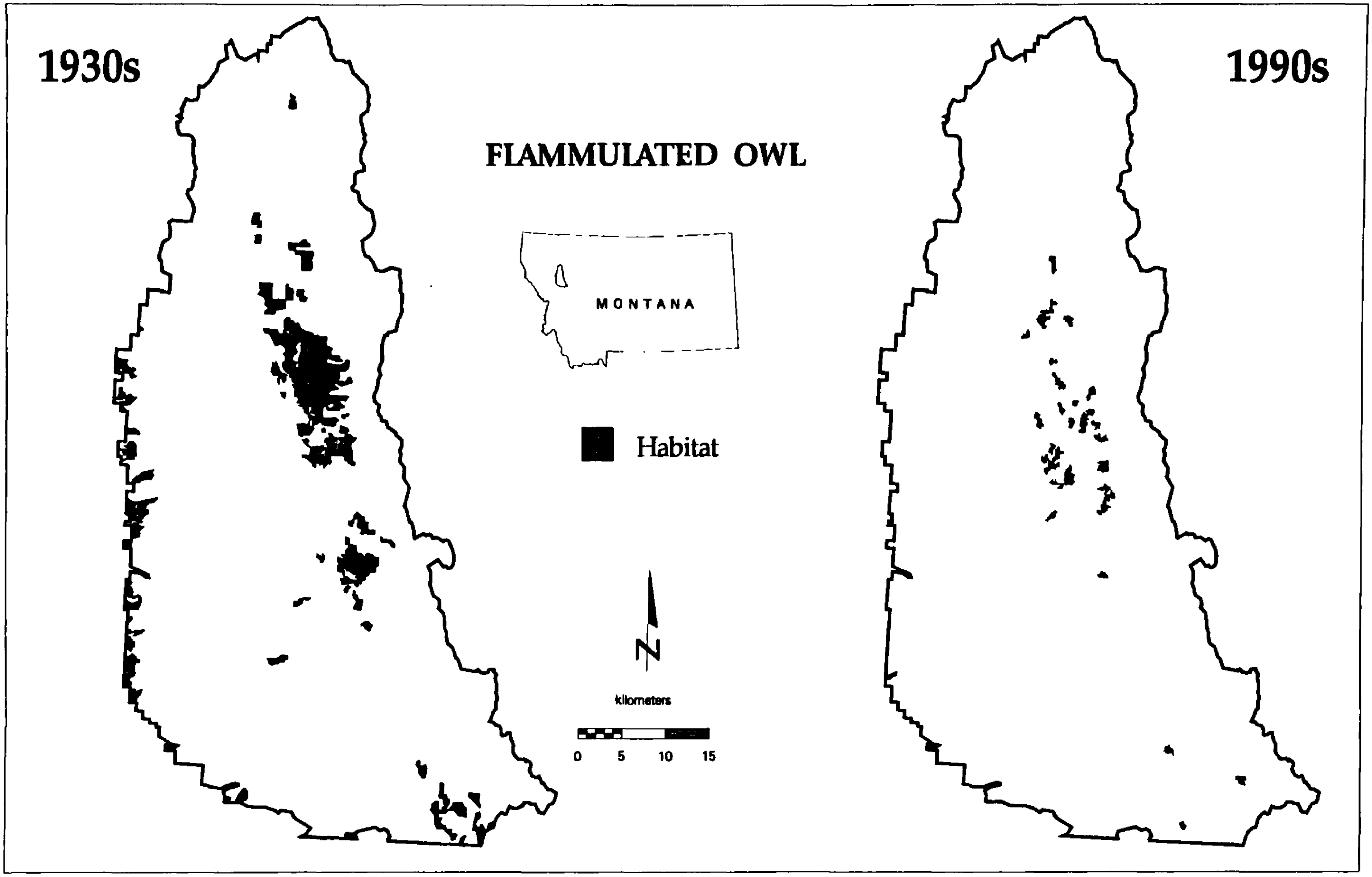


Figure 4-11. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the flammulated owl (*Otus flammeolus*).

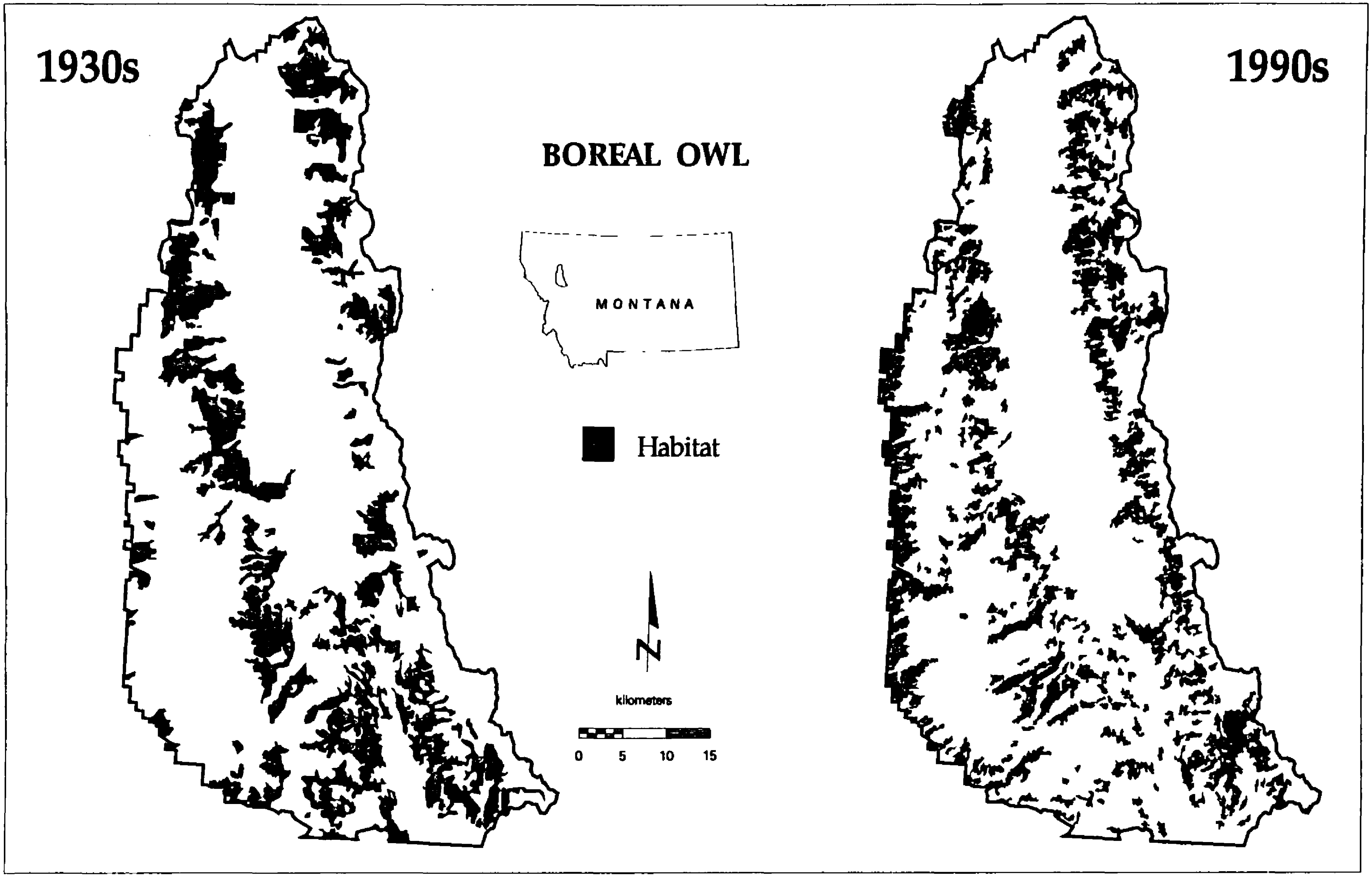


Figure 4-12. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the boreal owl (*Aegolius funereus*).

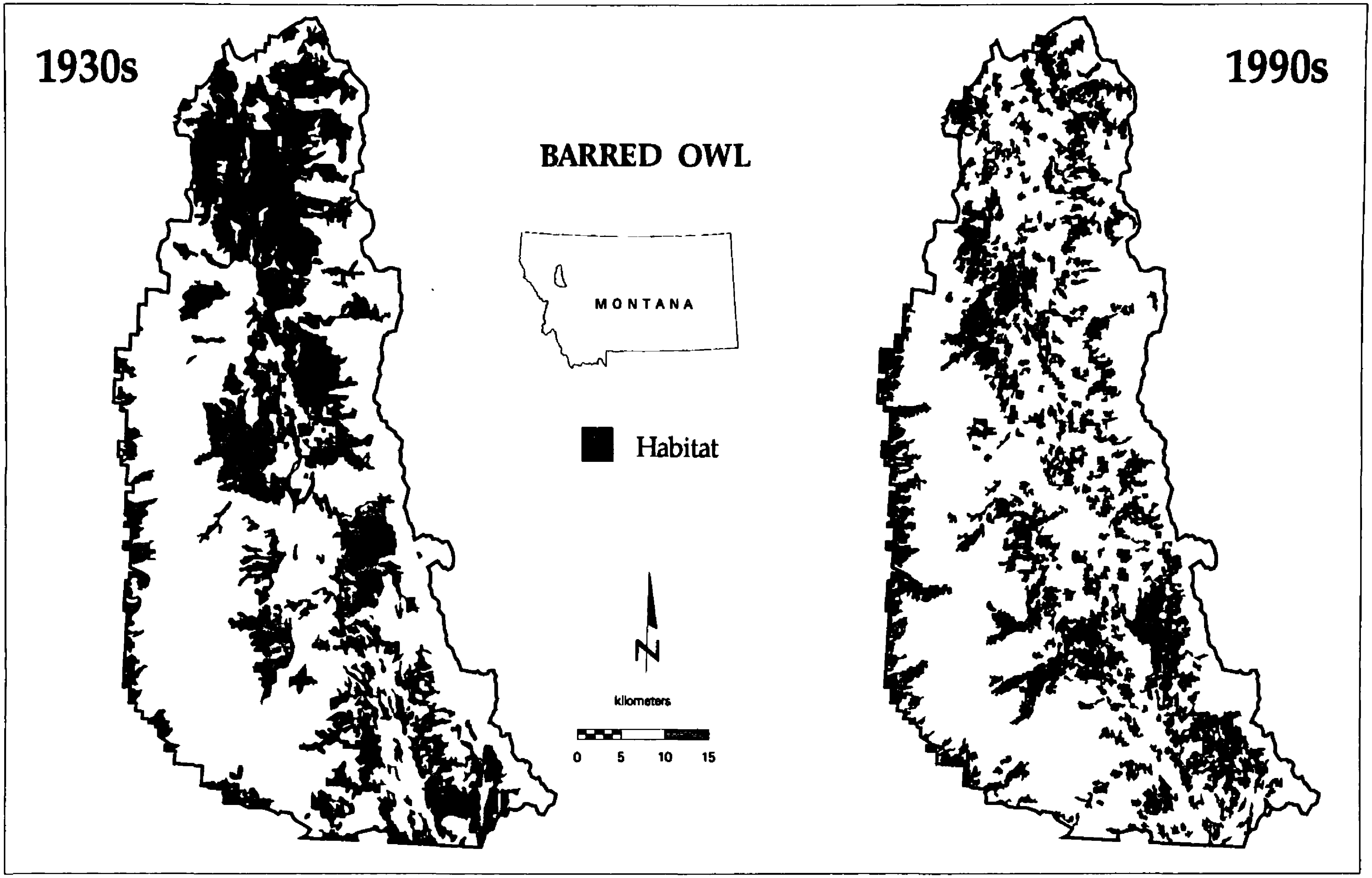


Figure 4-13. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the barred owl (*Strix varia*).

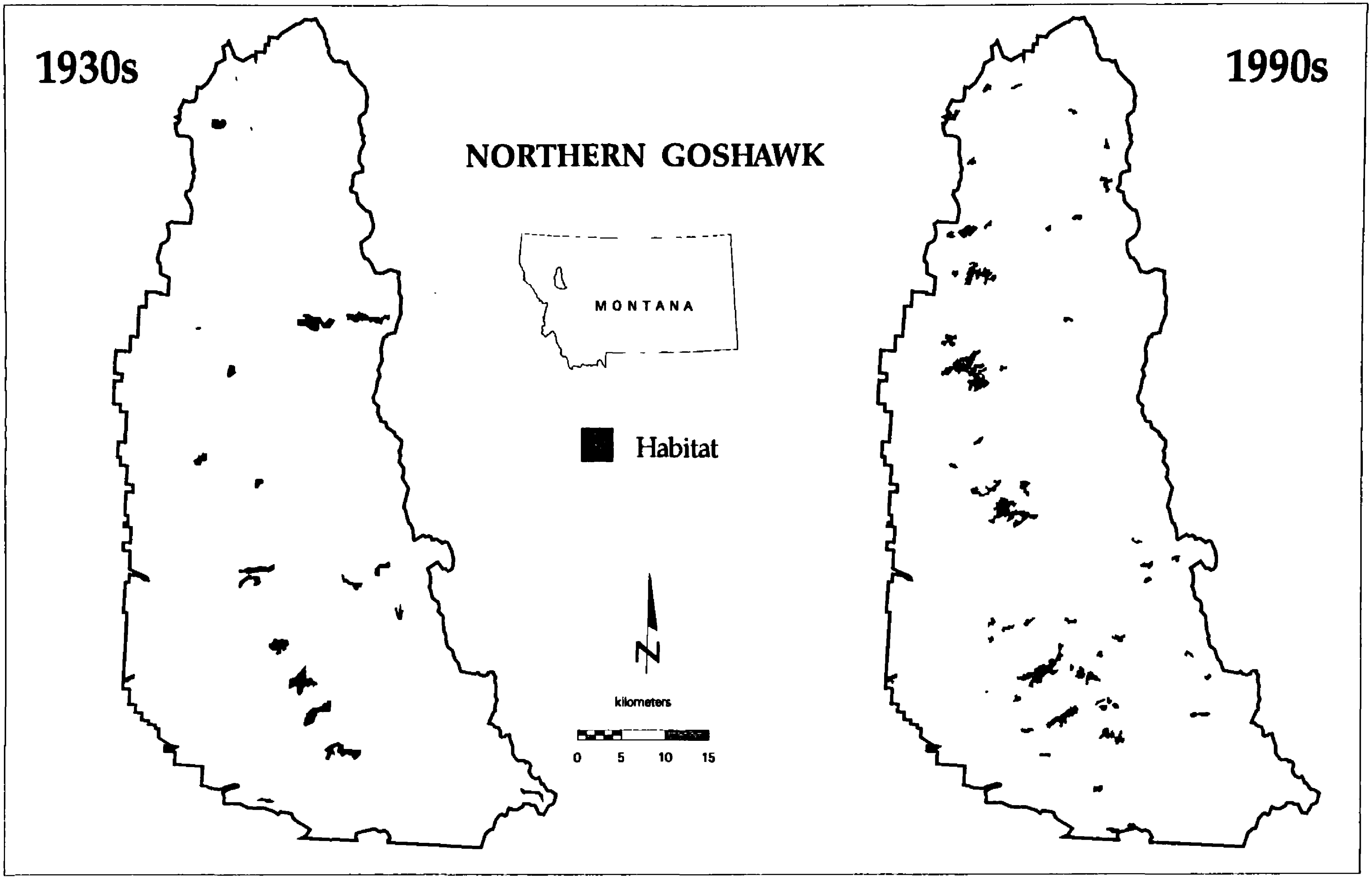


Figure 4-14. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the northern goshawk (*Accipiter gentilis*).

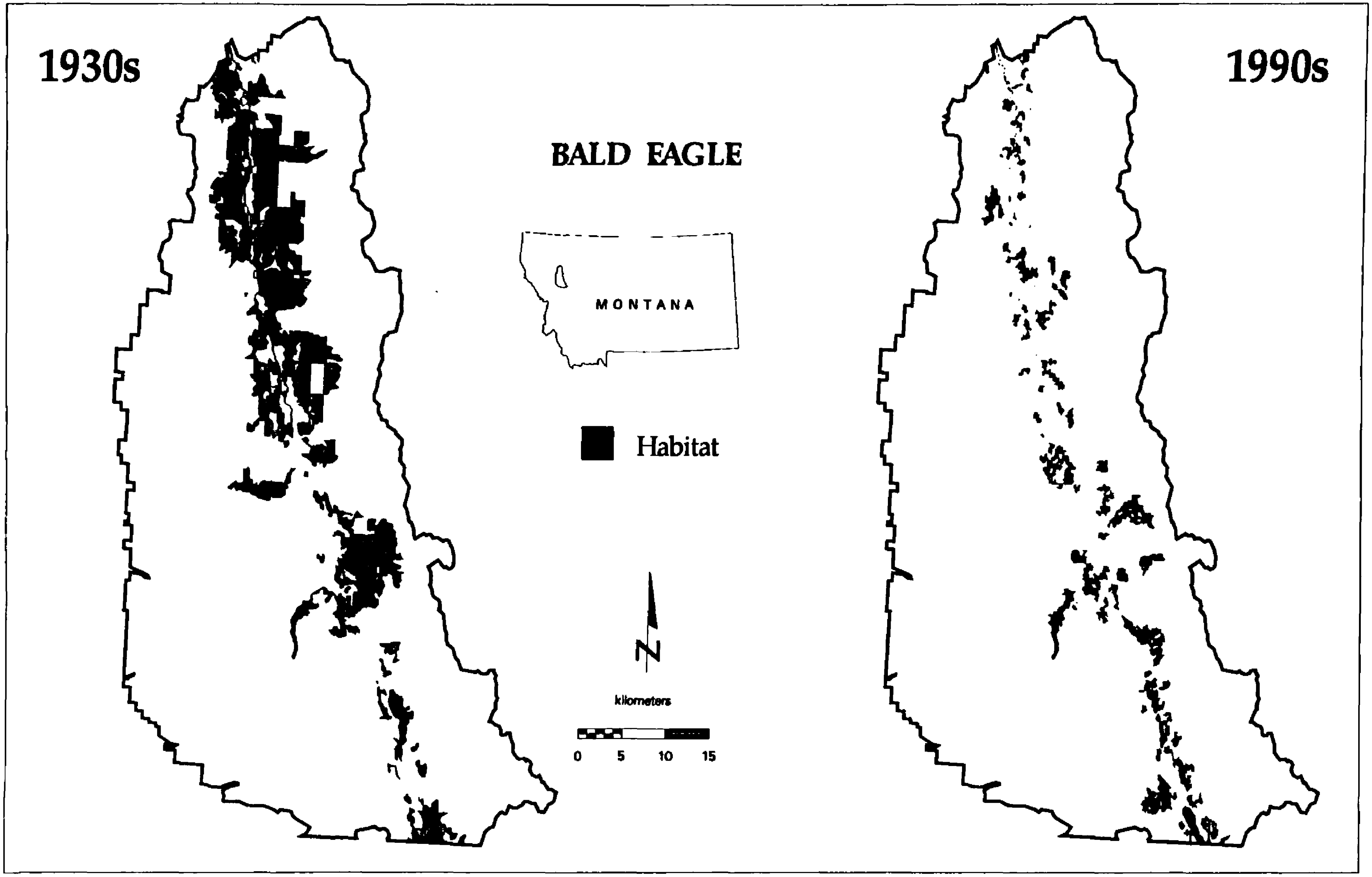


Figure 4-15. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the bald eagle (*Haliaeetus leucocephalus*).

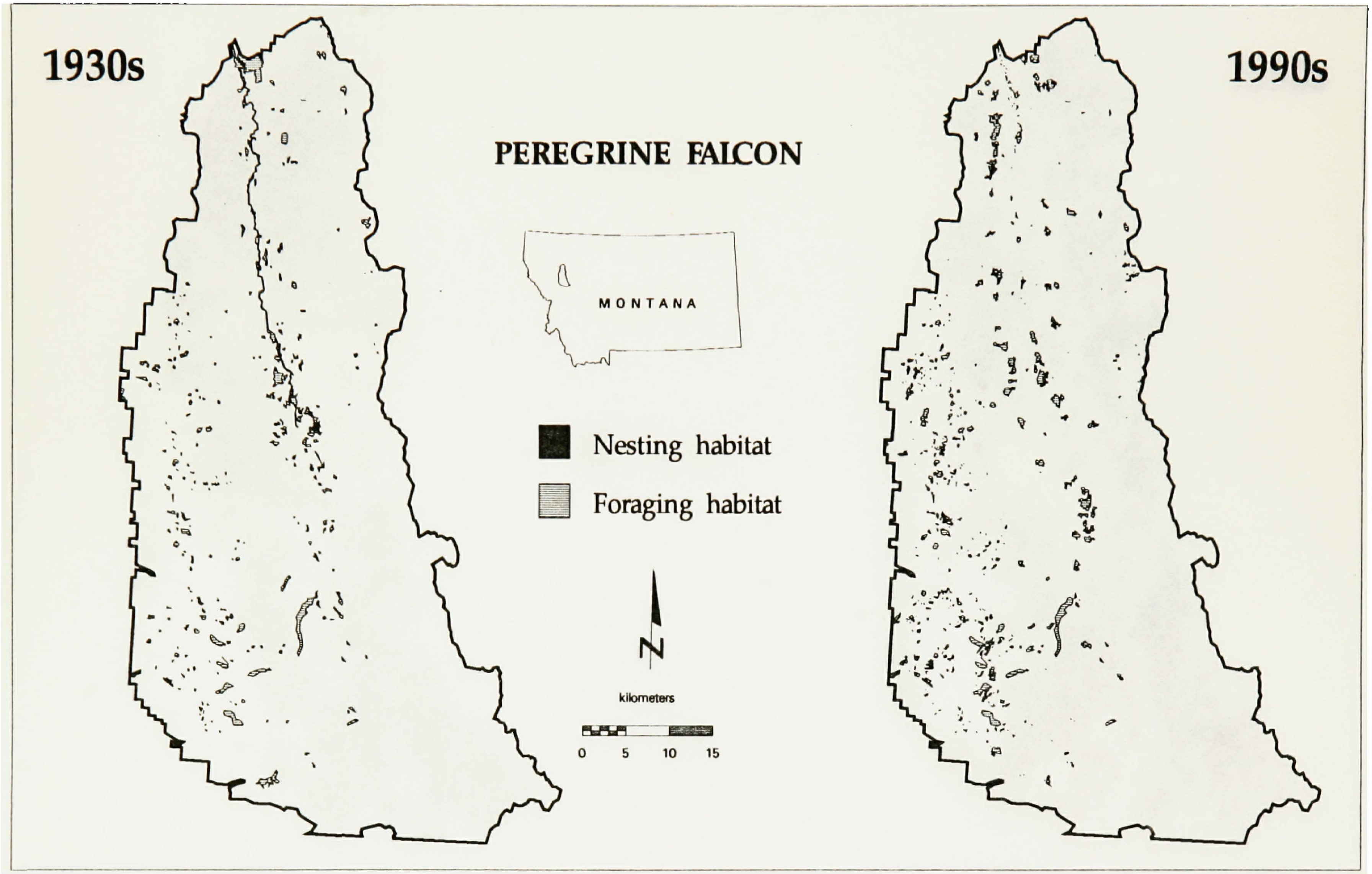


Figure 4-16. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the peregrine falcon (*Falco peregrinus anatum*).

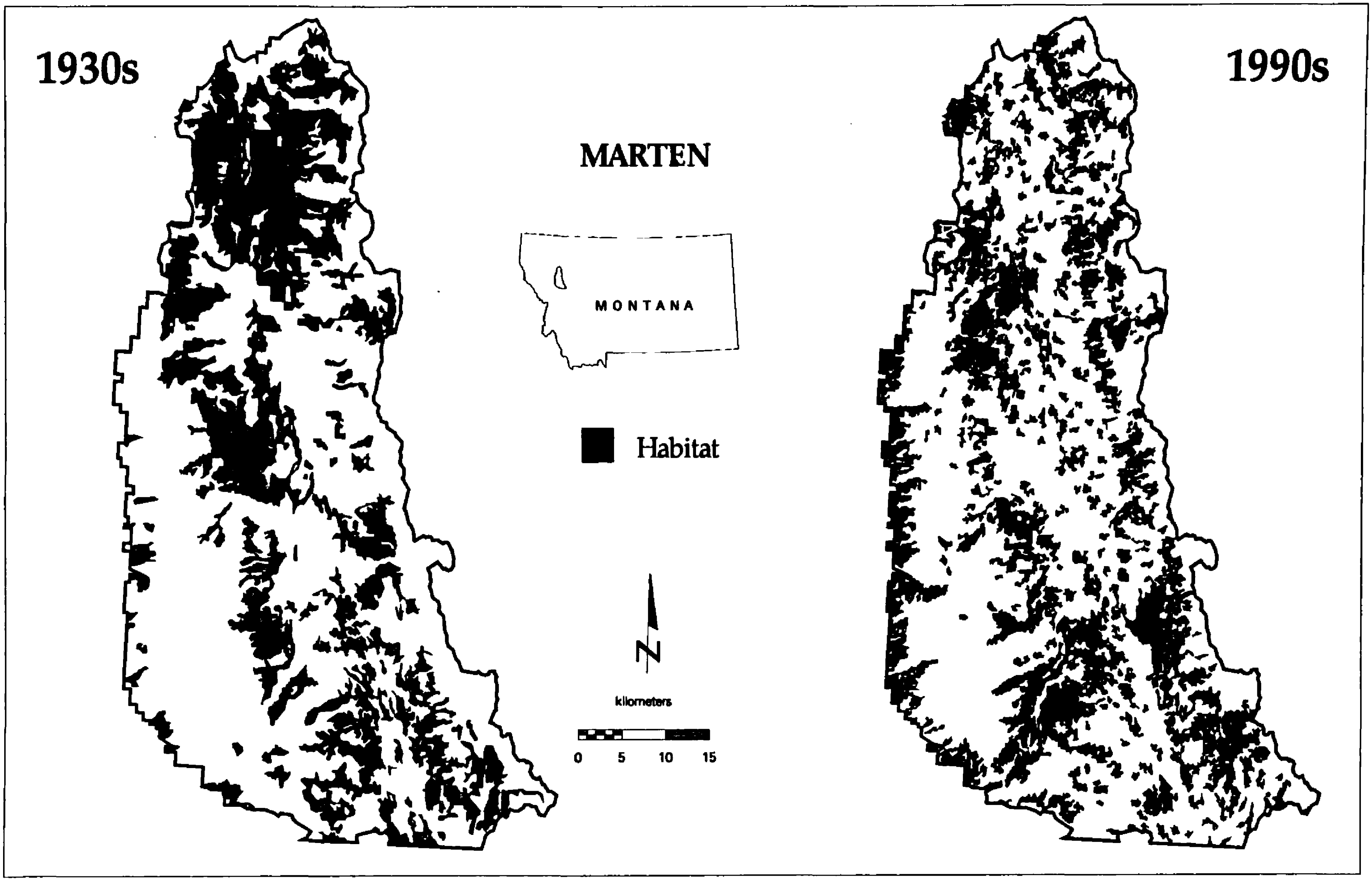


Figure 4-17. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the marten (*Martes americana*).

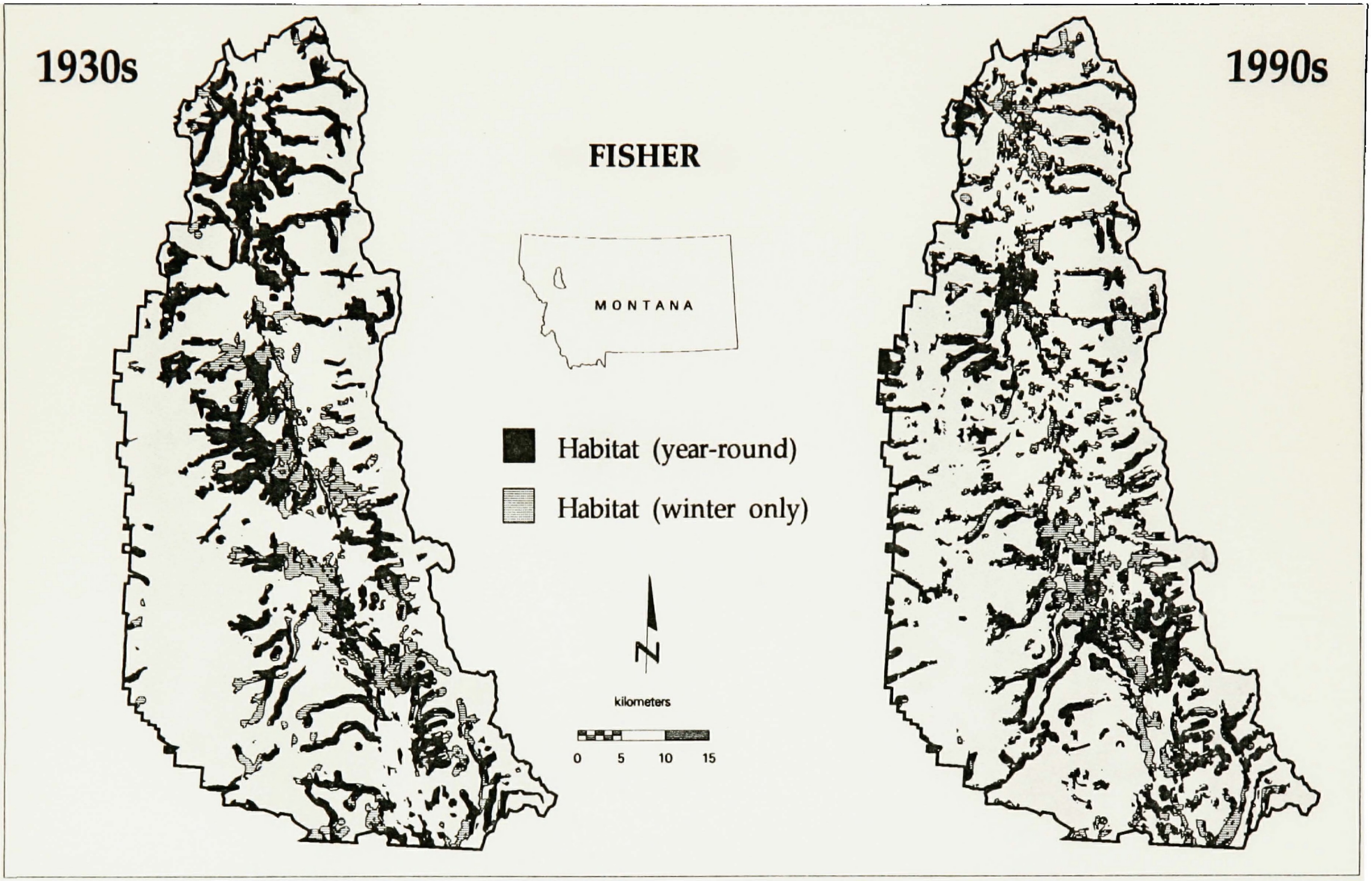


Figure 4-18. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the fisher (*Martes pennanti*).

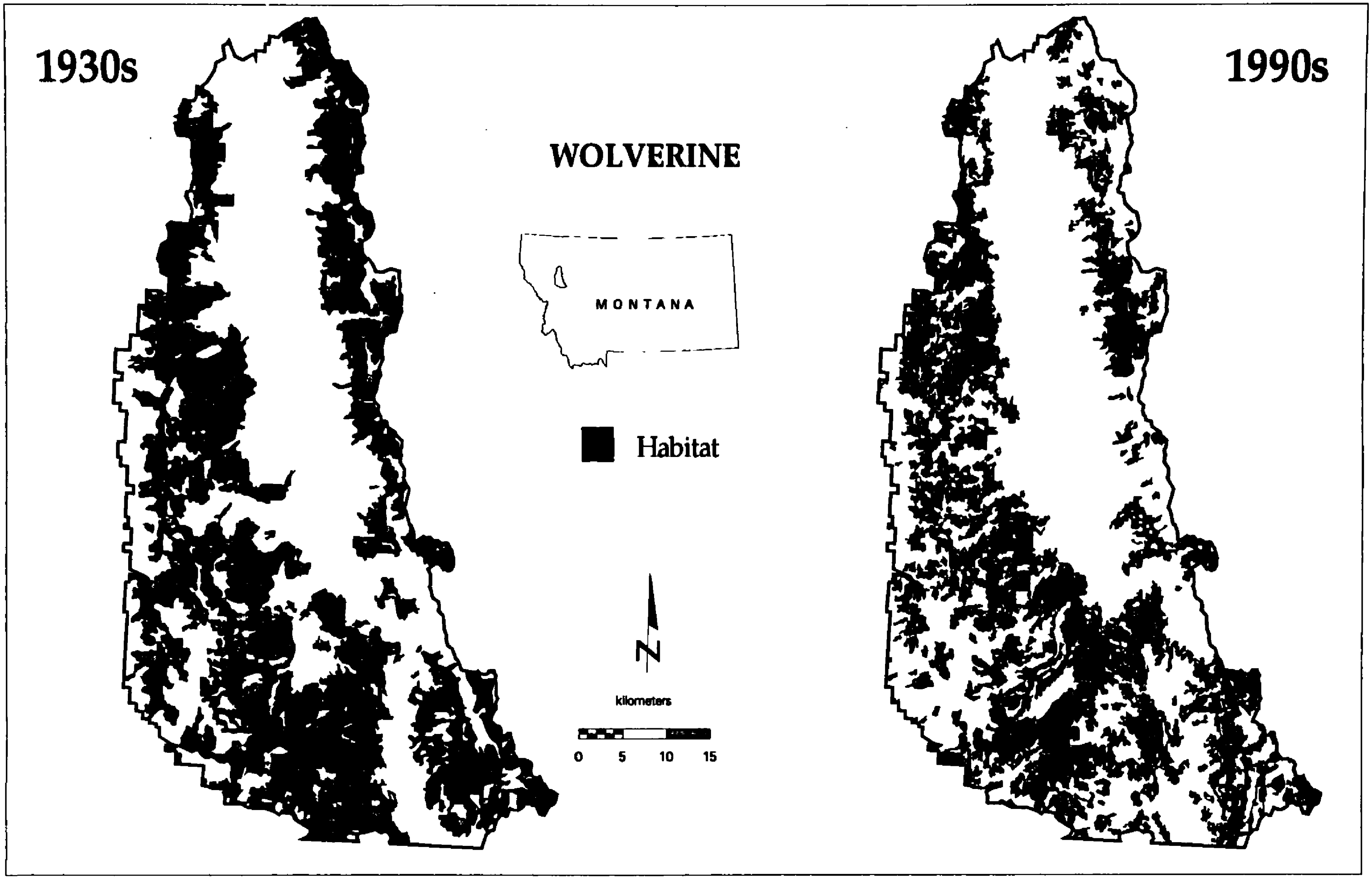


Figure 4-19. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the wolverine (*Gulo gulo*).

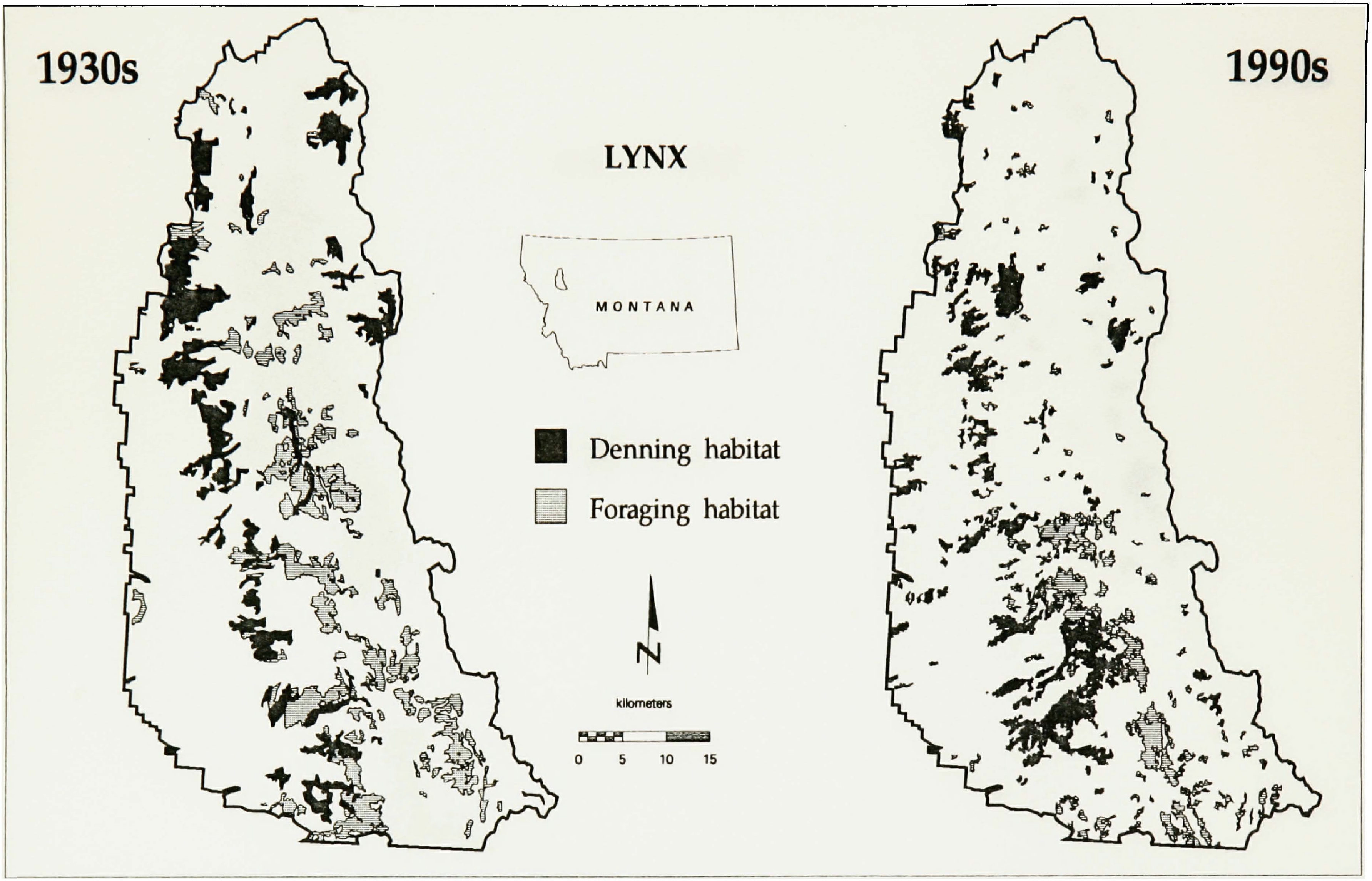


Figure 4-20. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the lynx (*Felis lynx*).

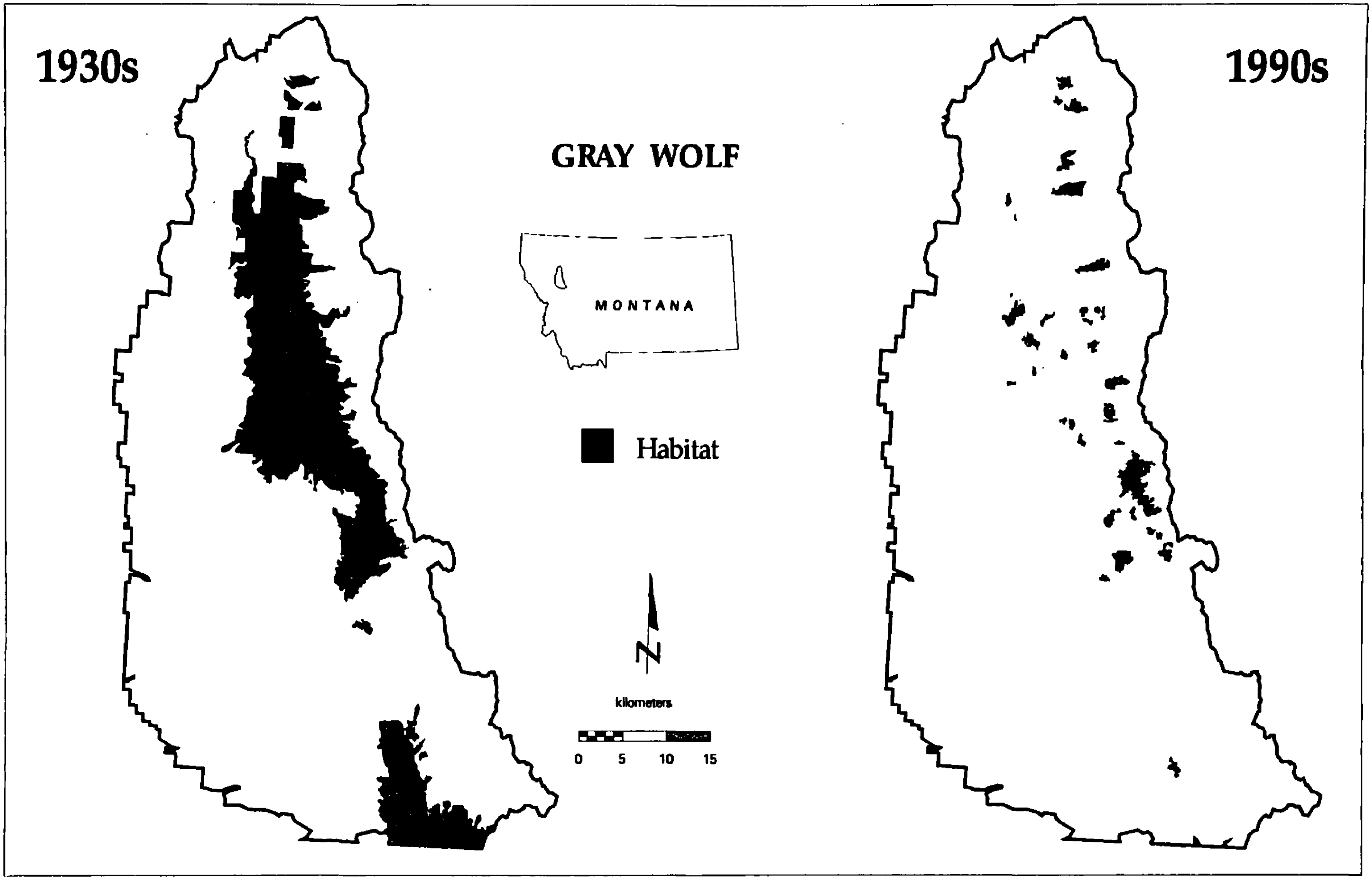


Figure 4-21. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the gray wolf (*Canis lupus*).

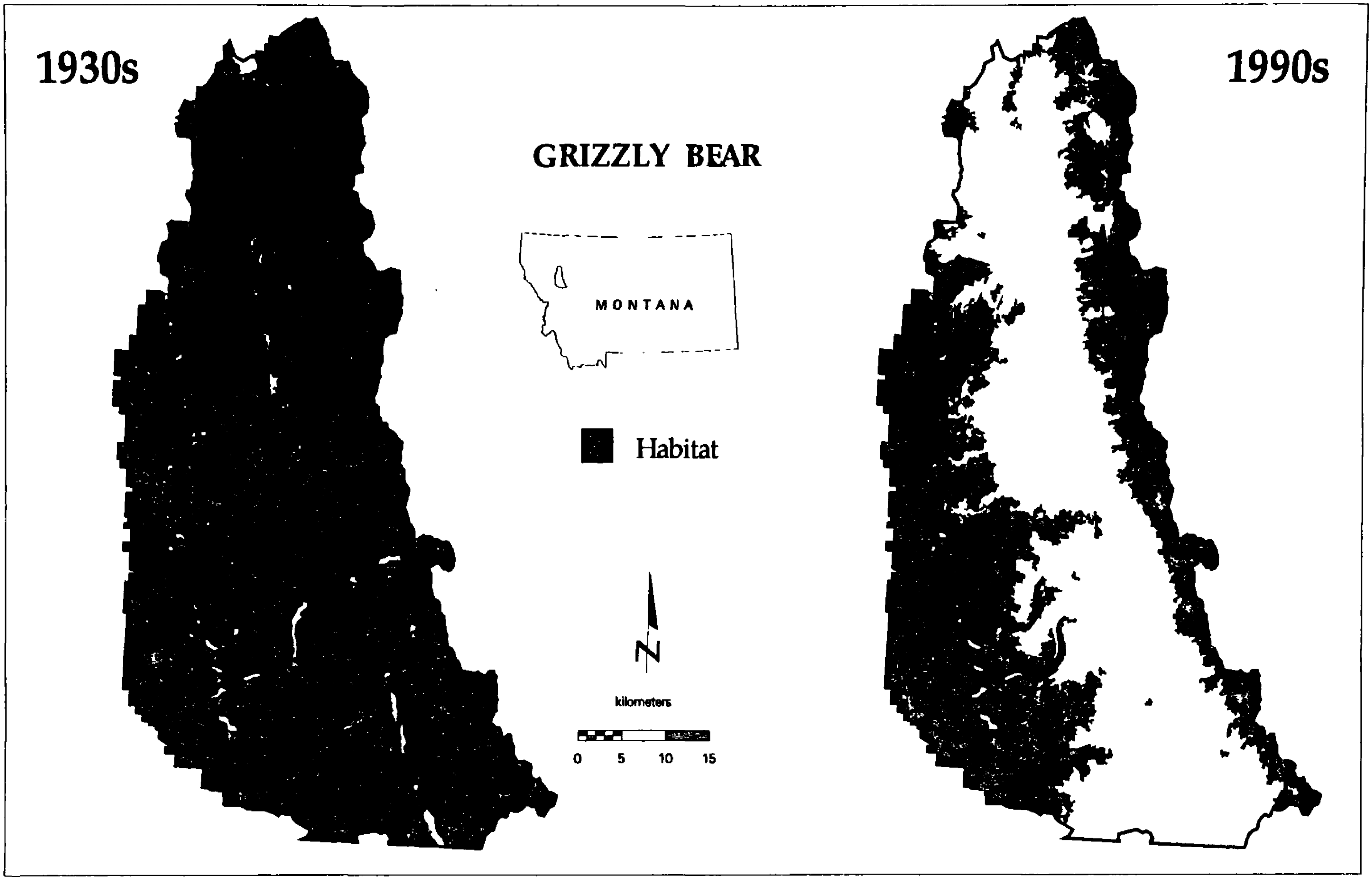


Figure 4-22. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the grizzly bear (*Ursus arctos*).

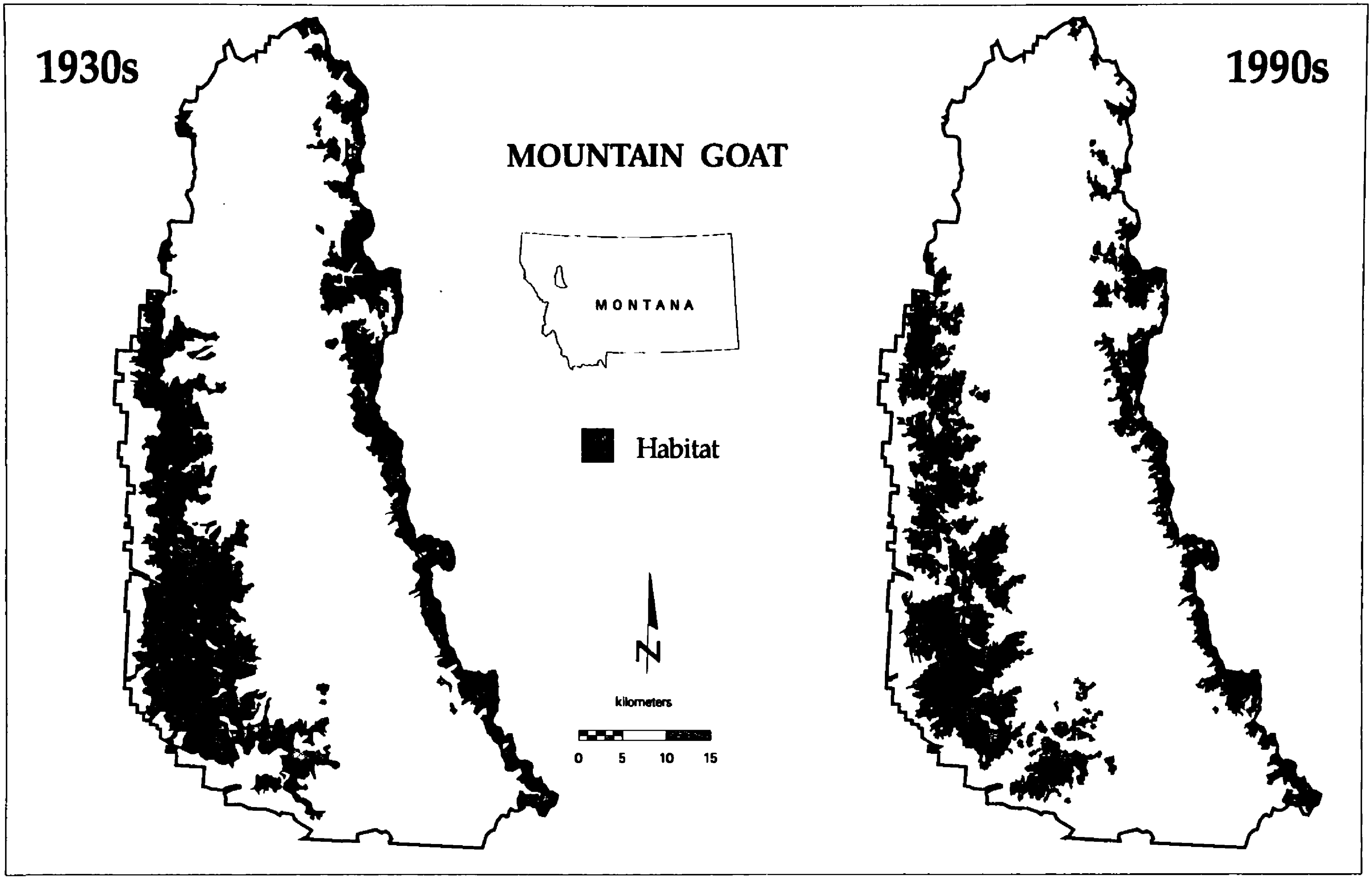


Figure 4-23. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the mountain goat (*Oreamnos americanus*).

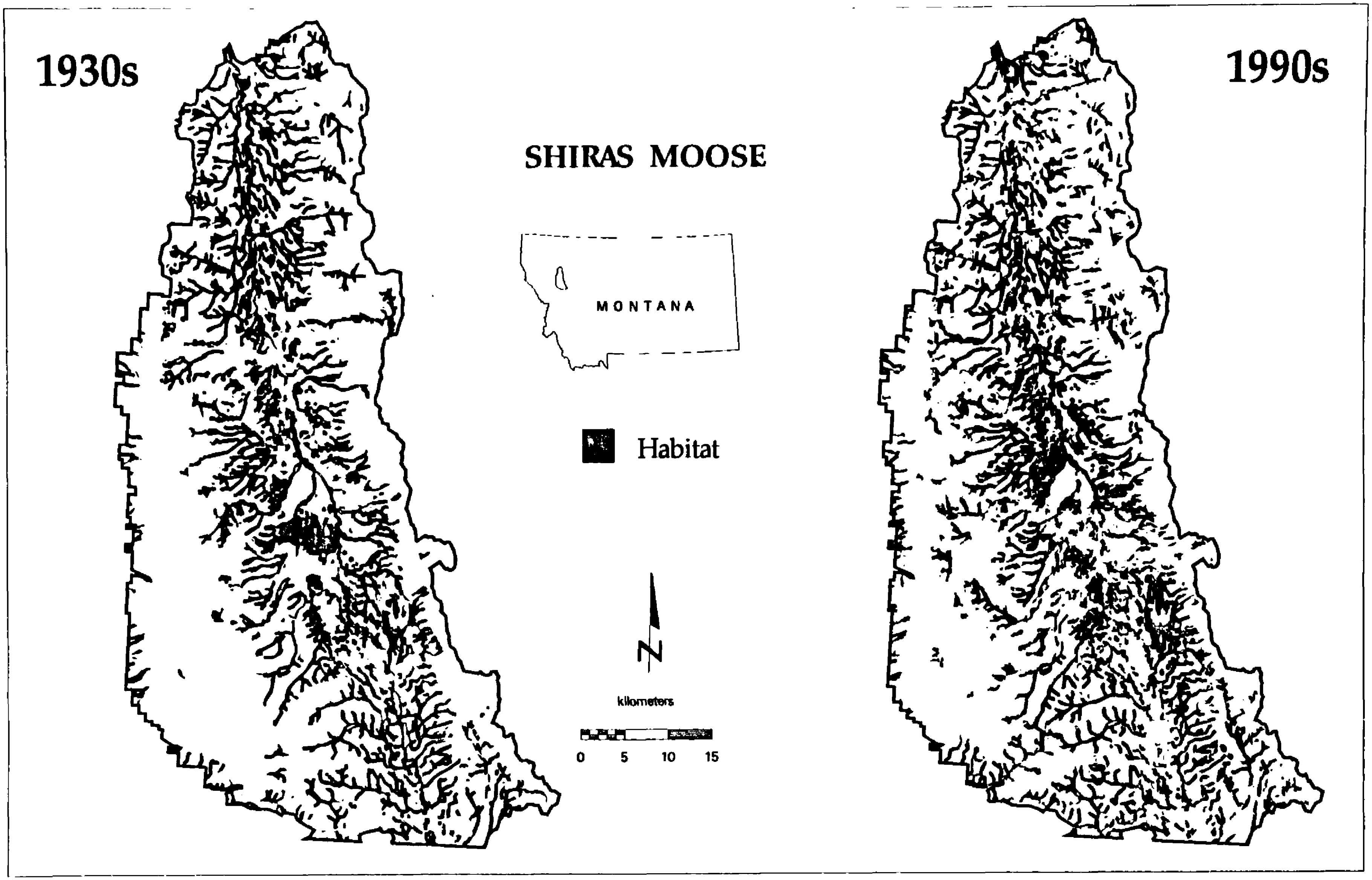


Figure 4-24. A comparison of predicted habitat (based on 16 ha MMU vegetation) in the Seeley-Swan landscape for the Shiras moose (*Alces alces shirasi*).

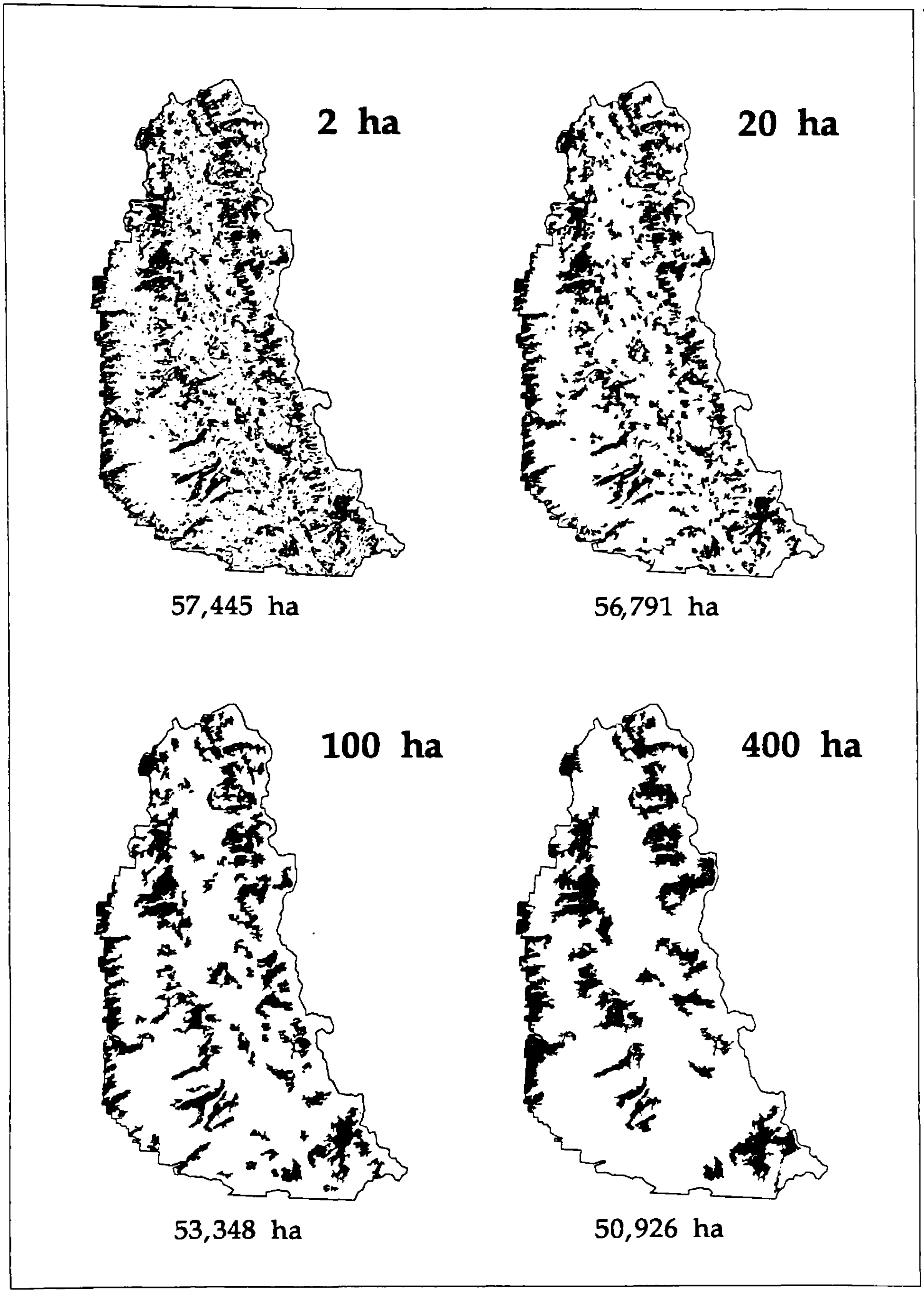


Figure 4-25. Predicted habitat for the pileated woodpecker (*Dryocopus pileatus*) at four different minimum mapping units, 1990s vegetation.

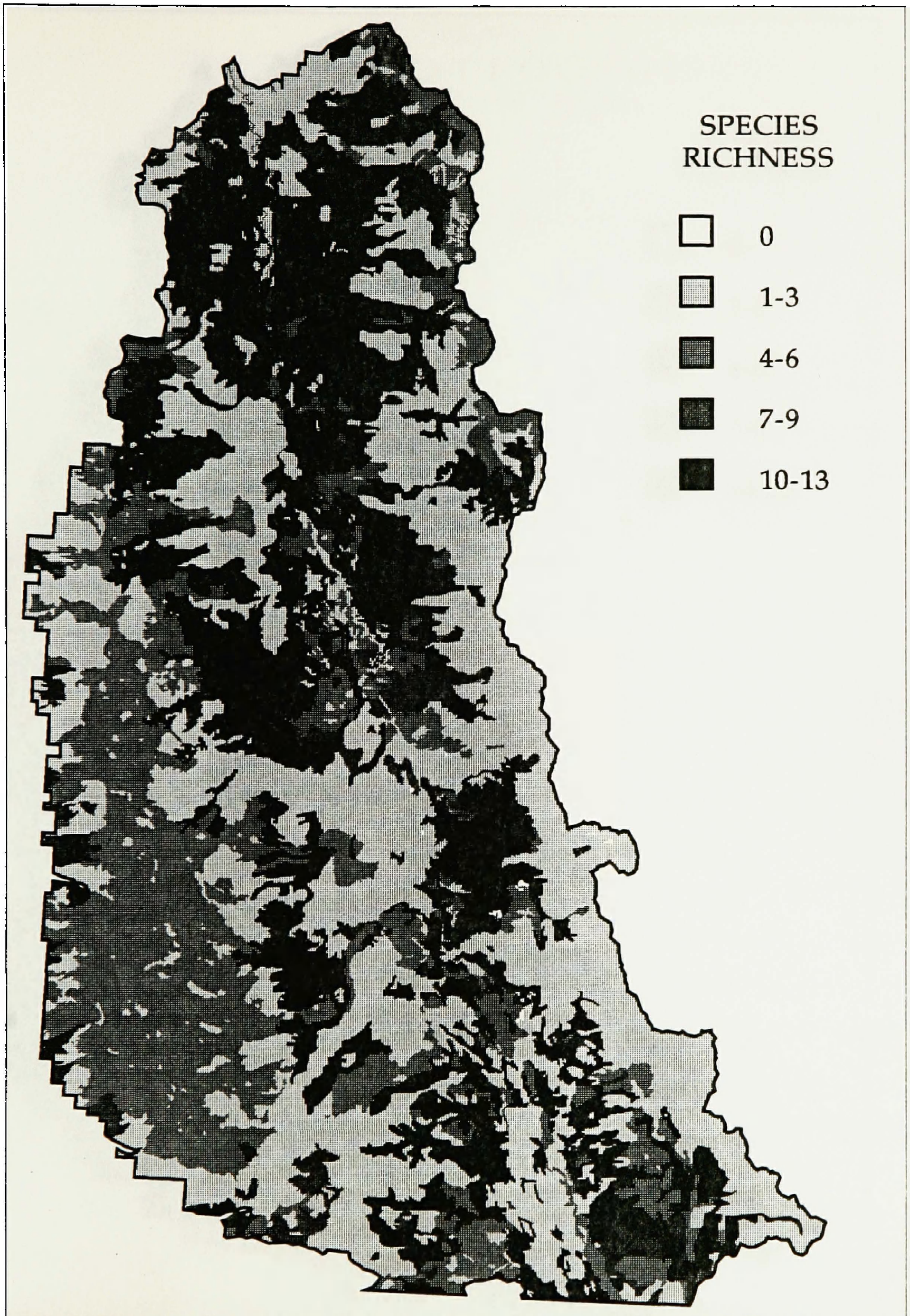


Figure 4-26. Number of species predicted to be present for each polygon in the Seeley-Swan landscape, 1930s. Habitat was modeled for 20 species altogether.

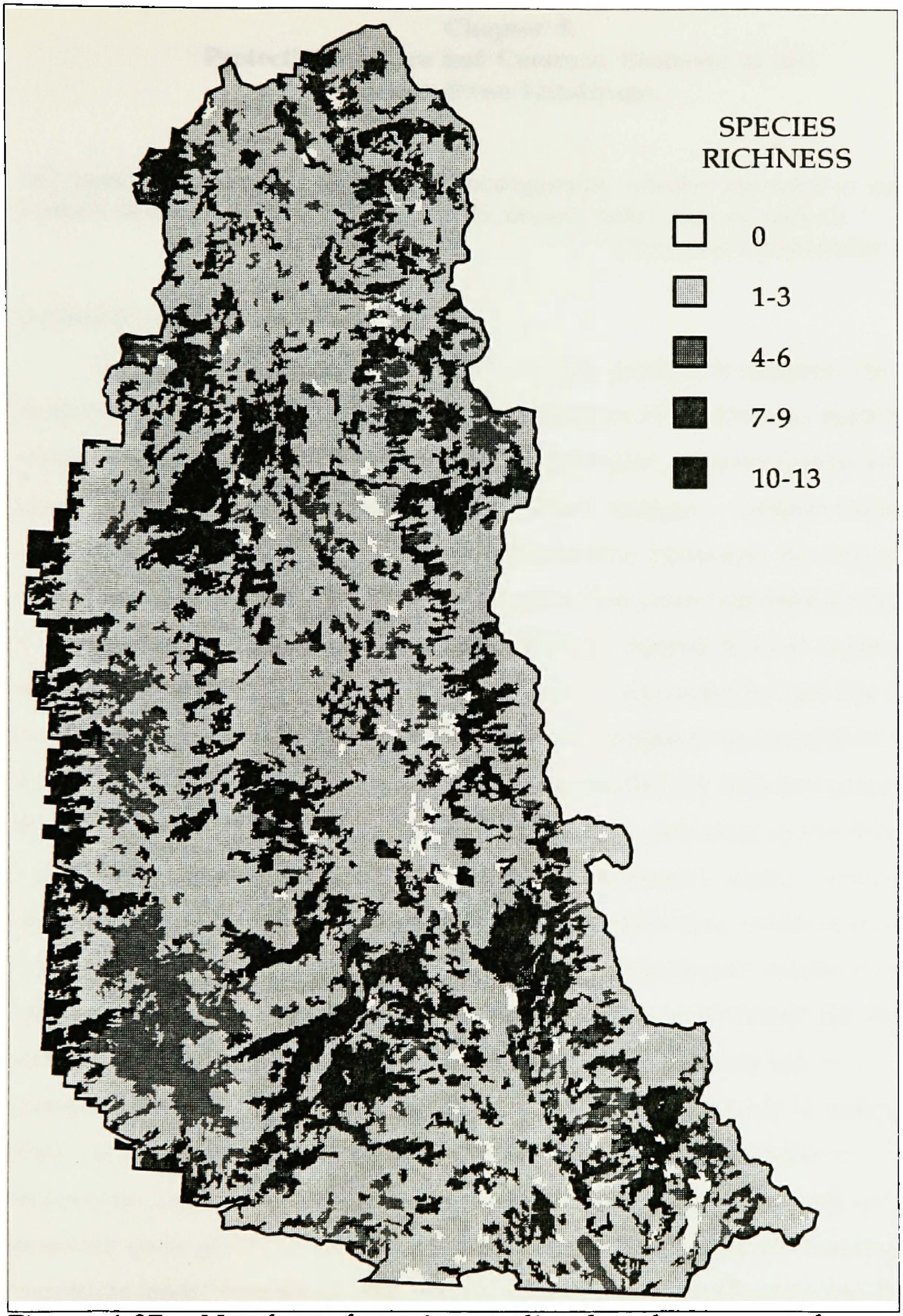


Figure 4-27. Number of species predicted to be present for each polygon in the Seeley-Swan landscape, 1990s. Habitat was modeled for 20 species altogether.

# Chapter 5:
# Protection of Rare and Common Elements in the Seeley-Swan Landscape.

*"All land management is biodiversity management, whether intended or not. ... It is much better to manage biodiversity by design rather than by default."*
*(Noss and Cooperrider 1994:28)*

## INTRODUCTION

Every human action has potential impacts, positive or negative, on biodiversity; thus, the ultimate goal of an evaluation of biodiversity must be to ensure its protection within a landscape. Traditionally, reserves (protected areas) have formed the backbone of sound conservation strategies. Despite increasing attention to management practices in the surrounding landscape, the design of reserve networks remains the most viable option for protecting biodiversity. Wilderness areas, Research Natural Areas (RNAs), Special Interest Areas, National Wildlife Refuges, and reserves privately owned by organizations like The Nature Conservancy provide examples of protected areas proposed or established in the Seeley-Swan landscape. Such reserves were designated for different purposes and are managed with different objectives. For example, according to the Wilderness Act of 1964, *wilderness areas* are dedicated to recreational, scenic, scientific, educational, conservation, and historical uses. By definition, wilderness areas are "untrammeled by man." They should be areas of undeveloped federal land where "the imprint of man's work" is negligible, outstanding opportunities for solitude or primitive recreation exist, area is sufficient for preservation and use in an unimpaired condition (5000 ac suggested minimum), and features of ecological or other value may be present (16 U.S.C.A. ss 1131(c)). The emphasis on recreational and scenic uses has perhaps led to the gorgeous "rock and ice" mountain vistas around which many wilderness areas are centered; however, the "worthless lands" hypothesis (see Pressey 1994 for review) suggests that many wilderness areas were designated because they were not suited for extractive resource uses. Thus, they may be skewed toward representation of inaccessible or less productive habitats. Nevertheless, wilderness areas protect vast areas and thus play important roles in maintaining landscape processes and supporting wide-ranging species. Unlike wilderness areas, *RNAs* are designated with more direct ecological objectives in mind: 1) to help preserve examples of all significant natural ecosystems for comparison with those influenced by humans; 2) to provide areas for education and research on the ecology, successional trends, and other aspects of the natural environment; and 3) to function as gene pools and preserves for rare and endangered plants and animals (Federal Committee on Ecological Reserves 1977). RNAs are primarily devoted to research and education, and are intended to provide baseline data for monitoring ecological changes. They may include typical or unusual flora, fauna, and/or other biotic phenomena, as well as characteristic or outstanding geologic, pedologic, or aquatic features and processes. Although the intent behind RNAs is laudable, such areas are typically quite small (the largest are <5000 ha, and 93% are <1000 ha (Noss 1990)), casting some doubt on the ability of some of these areas to maintain their natural character in the face of surrounding influences. *Special Interest Areas* are intended to protect unique landscape features with ecological and/or cultural values. They are administratively designated by the Forest Service, and contain scenic, geological, botanical, zoological, paleontological, historical, recreational, and other values meriting special recognition and management (Forest Service Manual 2372.05). Several botanical areas have been proposed in the Seeley-Swan. *National Wildlife Refuges* are also designated for ecological purposes, primarily protecting habitat of endangered species, perpetuating migratory bird populations, preserving natural diversity of all animals, and fostering understanding and appreciation of wildlife (Zaslowsky 1986). *The Nature Conservancy's* mission is to conserve biodiversity through establishment of natural area preserves, selected and designed to protect examples of as many native ecosystems and habitats as possible (Jenkins 1988).

Though more limited in scope, goals and objectives for the above reserves coincide reasonably well with general objectives for protection of biodiversity. To maintain the biodiversity of a region in perpetuity, Noss (1992) has suggested the following goals: 1) representing all native ecosystem types and seral stages across the natural range of vegetation in a network of protected areas; 2) maintaining viable populations of all native species in accordance with natural patterns of abundance and distribution; 3) maintaining critical ecological and evolutionary processes like disturbance regimes, hydrological and nutrient cycles, and biotic interactions; and 4) managing landscapes and communities to be responsive to environmental changes over short and long timeframes, and maintaining the biota's evolutionary potential.

Actually selecting sites for protection -- "the calculus of biodiversity," as it has been described by May (1994) -- may be the most critical task in meeting these goals. Major criteria for site selection include species richness, endemism, naturalness, rarity, area, threat of human interference, amenity value, educational value, scientific value, and representativeness (Margules and Usher 1981, Usher 1986); a site's role in maintaining natural landscape function should also be considered. The most important criterion may be representation (used synonymously with representativeness in this thesis, although Noss and Cooperrider (1994) draw a subtle distinction); based on the idea of designing a reserve network including every possible species (Margules et al. 1988), it corresponds to Leopold's (1953) goal of "keeping every cog and wheel" in a natural system. As a coarse-filter approach, vegetation has proven a suitable surrogate for other elements of biodiversity (Scott et al. 1993); in addition, representation of complete environmental gradients (such as elevation) is an important component of reserve selection (Noss and Cooperrider 1994). Finer filters can then be focused on rare and endemic species, or those known to be sensitive to human disturbance. To make the selection process more objective, iterative computer algorithms may be used to identify the smallest set of sites necessary to represent each species or community type the desired number of times (Margules et al. 1988, Margules 1989, Pressey and Nicholls 1989, Pressey et al. 1993). Although this approach may render site selection consistent and repeatable, it does not provide answers to the truly difficult questions in reserve selection and design: "Science cannot tell us precisely how many times or in what sized reserves each species or ecosystem type must be represented to be viable." (Noss and Cooperrider 1994:109) Thus, common sense dictates that conservation biologists prioritize sites according to irreplaceability, hot spots of richness and centers of endemism, poorest current representation in protected areas, and urgency of threat, then work to ensure that protection is afforded as far down the priority list as possible (Noss and Cooperrider 1994).

Despite its importance, site selection offers little insight into the design of reserves to protect chosen areas, including issues of area and connectivity (Noss and Cooperrider 1994). Thomas et al. (1990) offer five premises of reserve design: 1) Species that are well distributed across their range are less susceptible to extinction than species limited to small parts of their range; 2) large blocks of habitat containing sizeable populations of the focal species are preferable to small blocks with only a few individuals; 3) blocks of habitat that are closer together are superior to more separated blocks; 4) habitat in contiguous blocks is better than fragmented habitat; and 5) species are better able to disperse if the areas between blocks are more similar to the structural characteristics of the habitat blocks themselves. The most sensible approach to reserve design appears to be land-use zoning, allocating the most protection to a core set of sites, then surrounding sites with buffer zones to minimize the influence of nearby intensive land uses (UNESCO 1974, Harris 1984, Noss and Harris 1986, Noss 1987, Mladenoff et al. 1994). Adequate area and connectivity of the reserve network will play a critical role in the maintenance of normal landscape function, including disturbance regimes and movements of wide-ranging species, especially in the northern Rocky Mountains, a region with characteristically large and frequent disturbances, and a relatively full complement of animals with large area requirements.

Practical application of these basic principles of reserve selection and design to the Seeley-Swan landscape is my primary goal; however, analysis is directed primarily toward reserve selection, with more general recommendations made toward actual network design. My objectives are to describe patterns of ownership and management in the landscape; evaluate representation of vegetative types, elevation zones, and biophysical zones within the existing reserve network; and recommend sites for expansion of the network. Criteria for site selection include presence of cover types poorly represented in the existing network, elevation, forest seral stage (with preference accorded to mature/overmature stands), road density (as a measure of naturalness), and presence of sensitive plant species. Thus, both coarse- and fine-filter approaches are incorporated in the selection process. I will also address connectivity between protected areas and management of the surrounding multiple-use matrix for the conservation of biodiversity.

## METHODS

### Preparation of Base Data Layers

I first assembled and coded vector data layers for land ownership and management within the study area. Ownership boundaries were digitized from USGS 7.5' quadrangles (1:24,000 scale). Major bodies of water were mapped as areas without ownership. Data on management areas were acquired from the Flathead National Forest for Swan Lake Ranger District in raster format (50 m cell size), then converted to vector format. Section lines from the ownership layer were replaced into this vectorized file for consistency, and a spline function was used to smooth the stairstep effects of raster-to-vector conversion. The remainder of the study area was digitized from 1:24,000 mylar quadrangles provided by the Lolo National Forest. Management area boundaries had been transferred to these mylars from lines on a Forest Visitors Map (1:126,720), probably reducing map accuracy. Finally, the vectorized data from the Flathead National Forest were merged with the digitized data for Lolo National Forest to create a complete management layer. Polygons in this layer were then assigned an attribute for protection level based on the 9-level classification developed by the Montana Interagency Natural Areas Committee (1993, Table 5-1); major water bodies were maintained as a separate category. In this classification of protected areas, public lands are divided into six categories. Public lands with a high degree of protection for biodiversity values (i.e., maintained essentially in a natural state) and with secure designations are assigned to Level 111; lands with virtually the same protection but with designations which may be more easily changed are assigned to Level 112. Level 121 includes public lands with secure designations but with primary goals other than protection of biodiversity; some areas may be set aside to preserve given elements, and site manipulation to benefit those elements may be allowed. Level 122 includes a variety of agency-designated management areas, which may be changed at the regional or unit level of the agency, emphasizing certain components of biodiversity. Generally, resource extraction is not precluded, but it is often restricted in some way. Public lands suitable and available for resource extraction, and lacking designations for biodiversity are assigned to Level 131; because projects planned on these lands must comply with federal and state laws and regulations, such lands typically afford more protection than private, unprotected lands. Finally, public lands where the natural environment has been significantly altered on a long-term basis are assigned to Level 132. Private lands are divided into three categories: Privately-owned, formally designated nature preserves (Level 210), lands where some natural features are protected (Level 220), and lands with no explicit protection (Level 230). Levels 111 and 210 will be hereafter referred to as existing protected areas. National Forest lands were assigned to protection levels based on management area designations (Table 5-2), while State Forest lands were uniformly designated as level 131 (except for Goat Creek Headquarters -- level 132). All privately-owned lands were assigned to level 230 except preserves owned by The Nature

Conservancy (210) and lands under conservation easement (220). I also acquired data from the Montana Natural Heritage Program (September 1994), and summarized element occurrences (locations of species, community types, or other features or phenomena of interest, Jenkins 1988) within the Seeley-Swan landscape.

**Assessment of Representation**

Next, I assessed representation of landscape elements within current protection levels. I conducted assessments for cover types, elevational classes, and biophysical zones by overlaying those layers with a raster version of the protection layer and tallying the percentage of each type within each protection zone. The percentage of each type within existing protected areas was compared with the percentage of the type in the overall landscape as a measure of protection versus availability (a concept parallel to standard use/availability assessments of wildlife habitat selection). For cover type data, I used the standardized cover type/size class codes (Table 3-3) so that the percentage in each protection zone could be compared with the percentage in the landscape for both the 1930s and 1990s. Elevation classes were defined in 200 m contours ranging from $<1200$ m to $\geq 2800$ m.

To map biophysical zones (Table 5-3), I employed a method designed for use in the Columbia River Basin Assessment Area (Menakis, pers.comm.). Its basis is a 4 x 4 matrix representing temperature and moisture regimes (Fig. 5-1), and roughly corresponding to aggregations of habitat types (Pfister et al. 1977). Rules for the Seeley-Swan were drafted (Mantas and Sirucek, Flathead NF; Table 5-4) and implemented within ARC/INFO to generate a map of biophysical zones in the following manner. I first prepared three raster layers from the digital elevation model: 1) Elevation (500' contours), 2) aspect (north and east, 0-135° and 315-360°; south and west, all other aspects; and flat, 0-5% slope), and 3) slope gradient (0-5, 5-30, 30-60, and 60+). Next, new layers were built for each biophysical zone based on combinations of the above layers, as defined in Table 5-4; separate layers were generated for areas north and south of the Clearwater Divide. Finally, each of the biophysical zone layers were merged to generate one map of biophysical zones for the entire study area. The rules I adopted were designed for use at coarser resolution; given more time, I would attempt to refine the rules for use at 1:24,000 scale.

After underrepresented cover types were identified, their distribution, both in the landscape and in relation to specific topographic and biophysical types, was examined. Layers of underrepresented cover types were prepared, then overlayed with layers for biophysical zones and 200 m elevation classes. Results were graphically displayed. In addition, for all cover types, elevation ranges were examined.

Representation of wildlife habitat within existing reserves was also addressed; the percentage of predicted habitat in existing protection was calculated for each of the 20 species modeled in Chapter 4. As a fine-filter evaluation, representation of element occurrences was assessed by overlaying point locations for each occurrence with the raster protection layer and tallying the number of points in each protection level. Element occurrences were also mapped in relation to existing protected areas. Sensitive plants were grouped according to habitats where they are found (Table 5-5), and similarly mapped.

**Reserve Selection**

Next, I created a database for use in evaluating sites for potential inclusion in a reserve network. Criteria for selection focused on those types found to be underrepresented in the previously-described analyses. To simplify the selection process, underrepresented biophysical zones were not included; I assumed that consideration of elevations below 1600 m would adequately capture those biophysical zones. For each of 24,903 polygons in the 1990s vegetation layer, I created new attributes for underrepresented cover types, mature/overmature forest, elevation $\leq 1600$ m, species richness, mean road density, and presence of plant species of special concern. For the first three attributes, a value of 1 indicates presence of a type and 0 its absence; presence was identified through a series of queries on items already present in the database (cover type, size class, and elevation). Species richness is a summation of predicted presence or absence of habitat for the 20 wildlife species modeled in Chapter 4; although its theoretical maximum is 20, the highest observed value is 13. Mean road density was calculated by overlaying the vegetation polygons with a layer of total road densities (see Chapter 4 and Appendix B for methods) and for each polygon, averaging the road density values falling within its boundaries. I then assigned road density values to classes as follows: 0.00 = 0, 0.01-1.00 = 1, ... 11.01-12.00 = 12. I also added another 1/0 attribute for road densities $\leq$ or $>2$ mi/mi$^2$. This cutoff was chosen to represent areas used or avoided by grizzly bears (Mace and Manley 1993); certainly, areas with 0 road density are preferable sites for protection, but in the valley bottom where additional protection is most merited, 0 road densities are uncommon. As mentioned earlier, sensitive plant species found in the Seeley-Swan were classified based on habitat; attributes for each of five plant groups were assigned 1/0 values for each polygon based on presence or absence of plants in that group.

Next, I calculated scores for reserve potential by exploring various combinations of the above attributes. Scoring methods were similarly used in developing a conservation strategy for the Oregon Coast Range (Noss 1993) and in identifying linkage zones for grizzly bears in the Seeley-Swan (Servheen and Sandstrom 1993). First, I calculated scores as follows: Score1 = (Spp_Rich + Under_Cover + Under_Elev + Mature_OM + Plantgroup1 + Plantgroup2 + Plantgroup3 + Plantgroup4 + Plantgroup5) - Roadmean. The potential range of this score is -12 to 28; observed range was -11 to 15. Here, mean road densities played a disproportionate role, subtracting up to 12 points from the overall score. I thus calculated Score2 in the same manner, but without subtracting Roadmean; potential range was 0-28, observed range 0-15. Because the influence of roads is a factor in determining an area's naturalness, I wanted to include road density in scoring. Score3 was calculated in the same manner as Score1, but instead of subtracting Roadmean, I added another attribute (Roadscore, equaling 1 if mean road density $\leq 2$ mi/mi$^2$) to the equation, creating a potential range of 0-29 (observed range 0-16). All of the three scores above are heavily weighted toward mature/overmature forest types because 15 of the 20 wildlife models include at least a subset of mature/overmature forests. Thus, I calculated scores by a final, more equitable method: Score4 = Under_Cover + Under_Elev + Mature_OM + Roadscore + Plantgroup1 + Plantgroup2 + Plantgroup3 + Plantgroup4 + Plantgroup5; potential range 0-9, observed range 0-5. This last scoring scheme appeared most satisfactory and was used in subsequent analyses.

I examined a frequency distribution of scores for the 24,903 polygons, as well as the range of possible combinations resulting in individual scores. I then selected all polygons with scores $\geq$ 4 and created a raster layer to display their spatial distribution. These polygons are assumed to be the most diverse in terms of desired characteristics for potential reserves. To supplement diversity with a measure of rarity, I selected all polygons containing locations for sensitive plant species and created a raster layer to identify their distribution. Animal locations were not included because: 1) they were few in number, and 2) animals are less specifically tied to individual habitat patches, and thus their locations may be less reliable indicators of potential reserve value. Plant associations were not included because they were accounted for in assessment of representation of cover types. I next combined these diversity and rarity layers into one layer representing the highest-priority target areas for further evaluation. When combined with the existing protected areas, these target areas represent one alternative network of protected areas.

To explore a second alternative, I selected all polygons with a score of 3 based on the presence of underrepresented cover types, mature/overmature forest, and elevations $<$1600 m, assuming that high road densities might be excluding potentially valuable areas from consideration. This set of polygons was targeted to fill in spatial and compositional gaps observed in the first alternative. In addition, I created a 100 m buffer (200 m total width) around all perennial streams with the dual objective of including additional low-elevation habitat and enhancing landscape connectivity. Riparian buffers and polygons with score 3 were considered lower priority sites for inclusion in the network of protected areas. When combined with the higher priority sites above, as well as the existing protected areas, a second alternative network of protected areas emerged.

Next, I created data layers for both alternatives which showed the existing protected areas and first- and second-priority sites for augmentation of the network overlayed on current protection status. I identified major landowners and current management patterns for first- and second-priority sites. Finally, I reanalyzed representation of cover types, elevation zones, and biophysical zones for each of the alternatives, assuming that all sites identified in each alternative were assigned the highest protection status (Level 111). Spatial characteristics of the existing and alternative networks of protected areas were compared using FRAGSTATS (McGarigal and Marks 1994).

## RESULTS

### Ownership and Management

Land ownership in the Seeley-Swan is divided between federal, state, tribal, corporate, and small private holdings. The Flathead NF manages nearly 40% of the study area; when the Lolo NF to the south is included, National Forest lands account for over half of the study area (Fig. 5-2). Montana Department of State Lands holdings are mostly restricted to the Swan River State Forest in the northern part of the study area. All tribal lands are within the Mission Mountains Wilderness. While three-fourths of the Seeley-Swan is publicly owned, the land owner with the second largest area is Plum Creek Timber Company (20%). Corporate and small private lands are concentrated at lower elevations and distributed in a checkerboard pattern.

When lands are classified according to protection level (Fig. 5-3), multiple-use forest lands (Level 131) and lands accorded the highest protection status (Level 111) dominate the landscape, closely followed by private lands with no formal protective designation. Combined, these three categories occupy 82% of the Seeley-Swan landscape (Table 5-6). Almost all of the highly-protected public land is concentrated in the Mission Mountains Wilderness, the single largest patch in the landscape. Other protected areas -- including Research Natural Areas, candidate botanical areas, preserves owned by The Nature Conservancy, a USFWS refuge, and lands with conservation easements -- occupy only a tiny fraction of the landscape (<1%). Nonetheless, they are critical elements, and their spatial arrangement enhances the existing network of protected areas (Fig. 5-4, Table 5-7). Note, however, that the effective area of each protection level is highlighted by mean patch size and core area index (the proportion of a patch remaining when a 60 m buffer strip is subtracted from the inner perimeter): Smaller patches have less core area, and thus may be more vulnerable to outside influences. Mean patch size and core area are high for Level 111; this reflects the disproportionate influence of the Mission Mountains Wilderness, because most Level 111 patches are quite small (Tables 5-6, 5-7). Largest patch index, or the percentage of the landscape occupied by the largest patch of each level, also addresses effective area and contiguity. Aside from the large block occupied by the Mission Mountains Wilderness, the only protection levels with large contiguous areas are Levels 122, 131, and 230. Lands with lower protection levels are better connected as expressed by mean nearest-neighbor distances. They are also principally located in the valley bottom, foreshadowing the results of the representation assessment.

### Assessment of Representation

The proportion of each cover type within each protection level is shown in Figure 5-5. As would be expected from ownership and management patterns, cover types associated with higher elevations are better protected than those typically found at lower elevations. To account for a range of natural conditions, cover types were only labeled underrepresented if the percentage in existing protection was less than the percentage of the cover type in both the 1930s and 1990s landscapes (Fig. 5-6). Based on this comparison, the following cover types were identified as underrepresented: mature/overmature mixed conifer (13), mature/overmature ponderosa pine (17), pole lodgepole pine (18), pole western red cedar (20), and mature/overmature western red cedar (21). The spatial distribution of underrepresented cover types is mapped in Figure 5-7.

Elevation zones showed the most obvious correlation with protection levels. As elevation increased, the proportion of each 200 m zone in Level 111 protection increased, ranging from 2% at elevations <1200 m to 99% at elevations ≥2800 m (Fig. 5-8). Elevations <1600 m were found to be underrepresented in existing protected areas (Figs. 5-9, 5-10).

Biophysical zones, modeled to roughly correspond to aggregations of habitat type groups, are mapped in Figure 5-11. As with cover types, the biophysical zones found at lower elevations were less likely to have high proportions in existing protection (Fig. 5-12). Biophysical zones 6,10, and 11 were identified as underrepresented (Fig. 5-13).

The distribution of underrepresented cover types within biophysical zones and elevation zones highlights the correspondence between these three landscape variables (Fig. 5-14). All four underrepresented cover types are most common in biophysical zones 6,10, and 11, and are concentrated in zone 10, which most strongly lacks representation based on the percentage of the landscape it occupies. Underrepresented cover types are also most common at lower elevations: Although pole stands of lodgepole pine and mature/overmature mixed conifer stands are found at higher elevations, ponderosa pine and western red cedar are almost entirely restricted to elevations below 1600 m (Fig. 5-15).

Element occurrences from the Montana Natural Heritage Database include eight animal species (20 locations), 29 plant species (144 locations), two state champion trees, and two plant associations (Table 5-8). Most elements occur in the valley bottom (Fig. 5-16), and slightly over 30% of the elements are located in areas accorded the highest protection (levels 111 and 210, Table 5-9). Similar trends are observed for sensitive plant species grouped by habitat (Fig. 5-17); only sensitive plant locations were incorporated in reserve selection. Of the 20 wildlife species modeled, the majority have 15-35% of predicted habitat in existing protection (Table 5-10). Species with the least amount of protected habitat include the wolf, harlequin duck, bald eagle, and flammulated owl; habitat for these species is restricted to lower elevations. Predictably, species with habitat concentrated at higher elevations, such as the boreal owl, wolverine, mountain goat, and grizzly bear, have a higher proportion of predicted habitat in protected status. For most species, the amount of protected habitat increased significantly for each of the alternative networks.

**Reserve Selection**

Scores for evaluating reserve potential assigned equal weight to underrepresented cover types, mature/overmature forest, elevations $<1600$ m, road densities $\leq 2/mi/mi^2$, and presence of sensitive plants. Thirty-six combinations of these attributes were obtained in the scoring process (Table 5-11). In most of these combinations, underrepresented cover types, low elevations, mature/overmature forests, or low road density were involved; plant groups played a relatively minor role. Only 4.2% of the study area (592 polygons) received scores $\geq 4$. These polygons were scattered throughout the landscape at elevations $<1600$ m (Fig. 5-18).

Polygons with scores $\geq 4$ were targeted as high priority sites for protection, representing a diverse spectrum of characteristics identified as desirable for this study. To supplement the sites selected in this coarse-filter analysis, all polygons containing at least one sensitive plant location were selected to ensure that these rare landscape elements were not left unprotected (Fig. 5-19). Most locations were concentrated in or near areas already protected (Fig. 5-4), but the selected polygons expanded on these existing protected areas. In all, 3.5% of the total landscape area was selected because of the presence of sensitive plants. Polygons with scores $\geq 4$ or containing sensitive plant locations were considered first-priority sites for further evaluation and possible inclusion in an expanded network of protected areas.

Sites with lower priority were also identified, including polygons with a total score of 3 (based on presence of underrepresented cover types, mature/overmature forest, and elevations $< 1600$ m) and 100 m riparian buffers around perennial streams. 1566 polygons (6.3% of the total landscape area) received scores of 3 as described (Table 5-11), whereas the riparian buffers occupied nearly 10% of the total landscape area. Figure 5-20 shows the spatial distribution of all selected sites in relation to existing protected areas.

Two alternative networks of protected areas were evaluated; the first (Fig. 5-21) included only the highest priority sites (i.e., scores $\geq 4$ or presence of sensitive plants), and the second (Fig. 5-22) included lower priority sites as well (scores of 3 and riparian buffers). The first alternative would add 5% of the total landscape to existing protection; most of this additional area would come from Levels 131 and 230. In the second alternative, an additional 18% of the total landscape would be allocated to existing protection; again, most of this area would come from Levels 131 and 230, although the total area in Level 122 also would be reduced. Note that these alternatives are no more than rough drafts meant to be of assistance to managers in final site selection and reserve design. The fragmented patterns observed in these alternatives would not be desirable in a network of protected areas, and thus would require modification if actual reserve boundaries were later delineated.

Landscape statistics were compared for the existing network of protected areas and both proposed alternatives. However, the results were heavily influenced by methods used to create the data layers -- when selected vegetation polygons were replaced into the existing protection layer, with its administratively defined boundaries, many small fragments were generated. Thus, general trends rather than specific results are reported: For the two alternative networks, mean values for patch size, core area index, and nearest neighbor distance decreased while the number of patches increased.

Representation of cover types, elevation zones, and biophysical zones became increasingly equitable for alternative networks 1 and 2 (Figs. 5-23, 24, and 25): Most types that were overrepresented remained constant or decreased, and underrepresented types increased in the proportion of protected areas they occupied. Exceptions include increases in seedling/sapling representation, attributed to selection for sensitive plant locations (because the increase is the same for both options), and increases in pole mixed conifer, which may be accounted to riparian corridors. Ponderosa pine representation increased measurably only for the second alternative, probably because road densities for most of these small patches in the valley floor restricted many to scores of 3 at best. Not all underrepresented types were present in the alternative networks in proportion to their presence in the landscape, but definite improvements were made for the most poorly represented types.

Finally, current ownership and management patterns were examined for first and second priority sites separately (Figs. 5-26, 27; Table 5-12). When areas already protected are subtracted, first-priority sites occupy about 12,488 ha. Flathead NF manages 43% of this area; 64% of the lands under Flathead NF's jurisdiction are currently in Level 131 multiple-use management, and 36% are in Level 122 and thus already accorded some specific protection for biodiversity values. Plum Creek Timber Company owns 32% of the total first-priority area (roughly 4030 ha). Second-priority sites cover approximately 32,116 ha. Again, Flathead NF is responsible for the largest proportion of this area (33%), and its land is currently managed under Levels 131 (62%) and 122 (37%). Plum Creek Timber Company owns 27% of the total second-priority area (8675 ha). Lolo NF holds jurisdiction over 17% of second-priority sites, whereas only 2% of first-priority sites were identified on Lolo NF. Seventy-five percent of Lolo NF lands are under Level 131 management, and the remainder are in Levels 122 and 132 (concentrated use areas).

## DISCUSSION

### Assessment of Representation

In the Seeley-Swan landscape, as in many landscapes of the western United States, the lower elevations are intensively managed and higher elevations almost uniformly protected. Observed patterns of representation within the existing network of reserves are thus unsurprising: If the lower elevations are poorly represented, one would expect the same trend for associated cover types and biophysical zones. It should be noted that all elevation and biophysical zones are represented, however limited in area, in existing protected areas. The same is true for all cover types except mature/overmature broadleaf forest (virtually absent in the landscape as mapped), urban and agricultural lands, and recently burned areas. My definition of adequate representation, however, requires a type to be represented in the reserve network in proportion to its occurrence in the landscape. This definition is potentially problematic because it does not account for the total area of the reserve network; theoretically, a 100 ha network could represent all types, but in very small amounts. Nonetheless, in the absence of strong direction from the scientific community regarding the ideal proportion of a type to be protected, this definition provides a solid and conservative guideline. Once a reserve network has been established, it may be difficult to ensure balanced representation, as illustrated by the large amount of additional area that must be reserved before representative proportions are roughly equivalent to landscape proportions (Figs. 5-23, 24, and 25). Pressey (1994) has noted the distinct disadvantages of *ad hoc* designations for reserve design: The content of regional reserve networks may be biased, leaving some species, communities, or ecosystems unprotected, and the goal of representing regional biodiversity may be made more expensive, reducing the chances of protecting many elements of biodiversity. Thus, the Mission Mountains Wilderness is in some respects a mixed blessing. Although this reserve provides critical habitat and security areas for many species, as well as representation of higher-elevation types, its sheer area makes the reservation of additional large tracts in the valley bottom potentially more difficult.

Cover Types. Underrepresented cover types in the Seeley-Swan include mature/overmature mixed conifer, ponderosa pine, and western red cedar stands, and pole stands of lodgepole pine and western red cedar. Technically, recently burned areas are also underrepresented because they occupied about 3% of the 1930s landscape, and undoubtedly even higher percentages at more distant points in time. However, it seems illogical to locate a reserve simply to enclose a recent burn; it would be more reasonable to 1) design a reserve network large enough to absorb the effects of the Seeley-Swan's characteristic fire regime, and 2) allow for reestablishment of the natural fire regime, in conjunction with restoration efforts to mitigate the effects of decades of fire suppression and minimize the likelihood of stand-replacement fires in settled areas. For similar reasons, I opted not to highlight pole stands of lodgepole pine in the reserve selection process, assuming that a reserve network experiencing natural disturbances will be likely to include adequate amounts of seral lodgepole. Large contiguous blocks of mature/overmature mixed conifer forests with a heavy western larch/Douglas-fir component covered much of the valley bottom in the 1930s, and likely played a critical role in landscape function. The largest remnant stands at low elevations should be targeted for inclusion in the reserve network; these appear to be concentrated between Seeley Lake and Holland Lake (or just north), and along side drainages at the north end of the Swan Valley (Fig. 5-7). Western red cedar is an important forest type within the study area because it is at the eastern edge of its distribution and may be a relict of past climatic regimes. This type is almost certainly overrepresented in the vegetation layer, as outlined in Chapter 3, but its general distribution accords well with my field observations. Stands are most concentrated along the northern end of the east slope of the Mission Mountains, and a reserve might appropriately be located in that region. The type is limited enough that land managers will undoubtedly be able to provide locations for the best examples of western red cedar stands. Ponderosa pine stands are also limited in the study area; in addition, patches are typically quite small and stand composition has shifted toward Douglas-fir. The majority of ponderosa pine stands are in the Condon vicinity, including Simpson Pines Candidate Botanical Area, which is probably the largest remaining stand in the Swan Valley. Likely, once reserves are established, restoration work will be needed to return stands to a condition where low-intensity, high-frequency fires can maintain open, parklike characteristics. In fact, such a restoration project is in progress in a 120 acre ponderosa pine stand near Condon, involving cooperative efforts between Flathead NF, Montana Logging Association, and Montana Wilderness Association (Missoulian, 12 Oct. 1994). Such cooperative efforts are a crucial aspect of management for biodiversity in the Seeley-Swan.

Sensitive Plants. Sensitive plant locations were assigned heavy weight in the reserve selection process because they are unique elements within the Seeley-Swan landscape, and in a larger regional context as well. In particular, *Howellia aquatilis*, a species federally listed as threatened, deserves special attention because the Swan Valley is one of only two major population centers (USDA:FS 1994b). This annual aquatic species is found in wetlands such as ephemeral glacial pothole ponds. Its genetic and autecological attributes render it especially sensitive to disturbance and loss of habitat; large wetland complexes including abundant subpopulations and numerous ponds of varying depths would offer *H. aquatilis* the best long-term protection (Lesica 1992). Condon Creek Candidate Botanical Area is proposed to protect a major cluster of *H. aquatilis*, and the Flathead NF is currently amending its Forest Plan to incorporate goals, objectives, and standards for conservation and recovery of the species (Holtrop 1994).

In addition to *H. aquatilis*, 28 other sensitive plant species are found in the Swan Valley, many in association with fens and other riparian habitats. Taking a conservative approach, I identified all polygons containing rare plant locations as highest priority sites in the reserve selection process. Distinct clusters were evident with this approach; especially significant clusters are in the vicinity of Lindbergh Lake, Condon Creek, and the area just south of Swan Lake (Fig. 5-19). Most of the other clusters require the allocation of large areas to protect one or two plant locations. The long, large strip in the north-central Swan Valley is a prime example of this problem, and probably should not be seriously considered as a potential reserve. Sites should be examined individually in terms of the sensitivity and rarity of species present; in most cases, buffer zones may offer adequate protection.

Almost all sensitive plant locations are north of the Clearwater Divide. While this may reflect differential survey intensity, the southern part of the landscape has been surveyed, and the lack of sensitive plant locations is most likely a function of habitat differences (Evenden, pers. comm.). However, most sites with scores $\geq 4$ were also north of the Clearwater Divide, creating a potential gap in the spatial arrangement of reserves. For this reason, lower priority sites should be given more importance in the Seeley Lake part of the study area. In particular, old-growth stands of western larch in the Chain of Lakes area should be targeted for inclusion in a reserve network, despite management complications created by concentrated human use.

## Reserve Selection

I opted against iterative approaches to reserve selection (Margules et al. 1988, Margules 1989, Pressey and Nicholls 1989) because they are best utilized for very large study areas and highly complex data sets, where the sheer number of potential combinations can prove overwhelming. Despite their objectivity, such approaches seem slightly impersonal. Working with a relatively small landscape and prioritizing sites based on a handful of critical attributes, I found it more efficient to simply score all sites, then present the results for use in further, more subjective analyses. My prioritized sites can now be evaluated individually in terms of area, location, vulnerability, and contribution to the diversity of the reserve network. In a conservation strategy like this, subjectivity in the form of professional judgment can play an important role. Now that sites have been prioritized, those individuals with extensive knowledge of the landscape should select actual reserves and delineate core areas and buffer zones using the suggested alternatives as a starting point. Because of the numerous landowners and mixed ownership patterns in the Seeley-Swan, cooperative efforts will be especially important. Many of the targeted sites are on privately owned lands; after the highest quality sites have been identified, opportunities for land trades, acquisitions, and conservation easements should be carefully explored.

## Reserve Design

There are no simple recipes to be followed in the design of reserves and reserve networks, although basic tenets have been adopted with regard to reserve size, shape, and proximity (IUCN 1980, Wilcove et al. 1986, Thomas et al. 1990), and excellent practical guidelines have recently been provided by Noss and Cooperrider (1994). In particular, the "plea for bigness and multiplicity" made by Soule and Simberloff (1986) is often echoed in recommendations for reserve design. Primary questions to be addressed in any evaluation include: Have all the elements deemed important been adequately incorporated? Has sufficient area been set aside to maintain landscape patterns and processes, and to allow protected areas to maintain their natural character when subjected to outside influences? What management practices are employed in the surrounding matrix? And is the system sufficiently understood to provide knowledge on when to manage intensively and when to adopt a *laissez faire* approach?

Individual Reserves Within the Network. In expanding the network of protected areas for the Seeley-Swan, a first goal should be to select the best example of each underrepresented type, and to add further examples as opportunity permits. Although "best" is a subjective term, the actual criteria applied may be fairly objective. For example, in addition to reserve content (such as cover type, elevation, road density, and sensitive plant locations), reserve area, shape, and proximity may be evaluated. In this instance, larger stands would be favored, as would stands with greater proportions of interior habitat (low perimeter/area ratio) and stands best positioned to eliminate gaps in the spatial arrangement of the overall network. Actual reserve boundaries should be delineated according to natural gradients, like ridgelines or changes in vegetation, where feasible. In addition, buffer zones should be designated to help maintain the integrity of smaller reserves and to connect clusters of reserves. The concentric design of the multiple-use module (MUM) concept (Harris 1984, Noss and Harris 1986, Noss 1987), where protection is most intensive in core areas and use is most intensive in the outermost rings, may prove useful in integrating protected areas with the surrounding multiple-use landscape. Some modification to existing management areas will be necessary, but it should be noted that some management areas (those in Level 122 protection status) already function as buffers by protecting some biodiversity values. Efforts should be made to capitalize on these existing designations.

Area Considerations. Area may be the single most important factor in reserve design, especially in the northern Rocky Mountains, where fire regimes have historically played a predominant role in shaping landscape patterns (Arno

1980), and where large carnivores like the grizzly bear and gray wolf range over wide expanses. Maintenance of natural disturbance regimes should be a fundamental goal of reserve design (Baker 1992). Because fire has historically been the dominant disturbance regime in the Seeley-Swan, and is known to have affected broad areas over relatively short periods of time (see Chapter 3), presumably a network of protected areas adequate to support a natural disturbance regime would also be sufficient to maintain other processes, including biotic interactions and hydrological and nutrient cycles. But how should the adequacy of the network be defined? Consider that most landscapes are continually shifting mosaics of patches of different seral stages; before any patch can reach a stable state, disturbance typically intervenes (Sprugel 1991). A network of protected areas able to maintain the character of this shifting mosaic, with relatively constant proportions of the landscape in each seral stage over time, should be considered adequate for sustaining a natural fire regime (Noss and Cooperrider 1994). Fires in the Seeley-Swan historically affected extensive areas; thus, large reserves well-distributed throughout the study area would best be able to absorb natural disturbances. Because natural disturbance regimes and existing landscape conditions have been altered by half a century of fire suppression in the Seeley-Swan, restoration efforts will be necessary to return late-seral stands to the open, parklike condition historically common in the valley bottom.

Area considerations are also critical in the design of reserves to support populations of wide-ranging species, including the grizzly bear and wolf. By itself, the entire Seeley-Swan would be insufficient to support viable populations of these large carnivores; even the largest of western North America's national parks may be too small to ensure persistence of such species in the long term (Newmark 1987). Thus, linkage zones to facilitate movement between wildlands at a larger scale offer the most practical form of protection for these species (see below).

Connectivity. In general, connectivity between existing reserves should be strengthened in the Seeley-Swan; this can be most effectively accomplished through the establishment of riparian corridors. For demonstrative purposes, I placed 100 m buffers around all perennial streams, but this is neither a practical nor a defensible option: 1) these buffers occupy 10% of the total landscape area, in large part on privately owned lands; 2) existing guidelines in Forest Plans (USDA:FS 1985, 1986) and Montana Best Management Practices (BMPs, Logan and Clinch 1991) provide some protection for riparian habitats; and 3) 100 m is an arbitrary width, whereas a variable width fitted to individual riparian zones (and applied to intermittent streams as well) would be more appropriate if such buffers were implemented. Still, riparian corridors on perennial streams offer the best opportunity for north-south and east-west movements of species within the landscape. Although the specific merits of linkage zones have been heatedly debated, the need to maintain connections between populations is not disputed (see Noss and Cooperrider 1994). In addition to enhancing connectivity, riparian areas also provide habitat for many wildlife species of special concern, including the bald eagle, harlequin duck, and fisher. Note the dramatic increase in total protected habitat for many species under the second alternative, which includes riparian buffers (Table 5-10). Obviously, aquatic species stand to benefit greatly from riparian buffers as well. Public and private landowners should be encouraged to expand on existing guidelines for riparian management where feasible, thus ensuring maintenance of viable strips of habitat well-distributed throughout the landscape.

Connectivity between the Mission Mountains and Bob Marshall Wilderness Areas has also been addressed by Servheen and Sandstrom (1993). Although their analysis focused on grizzly bears, other species, especially forest carnivores, very likely would benefit from the linkage zones which they delineated (Fig. 5-28). Furthermore, these linkage zones, extending to the wilderness boundaries, represent a complete elevational gradient, and thus may play an important role in maintaining landscape function. Hence, reserve selection should be targeted toward these areas as well. Grizzly linkage zones were thus examined in relation to prioritized sites (Table 5-13). In all, linkage zones contained 59% of the area selected for sensitive plant locations, 42% of the area with scores $\geq 4$, 44% of the area with score 3, and 37% of the area in riparian buffers. The Clearwater Divide linkage zone (farthest south) protects the most priority area overall, but it is also by far the largest linkage zone. When size of linkage zone is considered, the Condon linkage zone (second farthest south) ranks highest in inclusion of prioritized areas, and the Clearwater Divide linkage zone ranks lowest of the four. Although not all critical sites for inclusion in the reserve network are located within linkage zones, areas of overlap between grizzly linkage zones and prioritized sites present an ideal opportunity for cooperative efforts. Sites containing important landscape elements can be protected while connectivity is maintained for grizzly bears and other wide-ranging species in the Seeley-Swan, thus helping to ensure population viability of those species in a larger regional context.

Regional Context. Viewed from a regional perspective, the Seeley-Swan landscape is situated between the humanized landscape of the Mission Valley to the west and one of the largest wilderness tracts in the Lower 48 states, the Bob Marshall, to the east. The Seeley-Swan, in its current semi-natural state, thus provides an important buffer zone for this extensive wildlands complex. In addition, when the focus is shifted to include a broader area, the Seeley-Swan's mesic low-elevation forests, wetland complexes, and concentrations of sensitive plants are seen to be unique landscape elements worthy of protection in their own right. Because almost 30% of the Seeley-Swan is already protected, the addition of very extensive tracts of land to the reserve network is not probable. Instead, features poorly represented in the current network should be protected in reserves large enough to remain viable in the face of outside influences, including edge effects on microclimate and habitat conditions, invasion of exotic species, and intensive land uses (see Janzen 1986). Further, some fairly large tracts of late seral and old-growth forest should be protected. Important in their own right as representative forest types offering a full complement of processes, such areas may also be useful as stepping stones throughout the valley bottom, connecting the Mission Mountains and Bob Marshall Wildernesses and providing habitat for wildlife species dependent on older forests. Not all of these areas need to be accorded the highest protection status; rather, some could be placed in long rotation cycles, and their locations in the landscape could shift over time. A *laissez faire* approach to reserve management will not be effective in this highly modified landscape: In some reserves, and in the surrounding matrix, restoration may be needed in the form of road obliteration, control of exotic species, and thinning and prescribed burning to restore processes which support open old-growth stands. Management practices within the matrix surrounding protected areas are also of critical importance. Application of New Forestry principles (Swanson and Franklin 1992) within the context of adaptive management (Holling 1978) will allow monitoring of success in conserving biodiversity on intensively managed lands.

**Landscape Indicators of Biodiversity**

Two primary indicators of high biodiversity values were identified in this analysis of the Seeley-Swan landscape: mature/overmature forests and riparian habitats. Older forests offer habitat for most of the wildlife species considered in this study. Thus, adequate protection of older forests will help ensure persistence of these species, many of which are accorded high management priority. Riparian habitats, particularly fens, harbor numerous rare plant species in the Seeley-Swan; western red cedar is also found in riparian areas. In general, diversity of vascular plants is high in riparian areas, and many animals use these habitats as well (see Naiman et al. 1993). Because these are typically small linear features, they can be difficult to identify using Landsat TM imagery. Their importance, however, makes them worthy of extra effort, and their presence can be modeled or inferred from other data layers, including hydrography and topography. A finer-resolution assessment of riparian habitats could only improve this study.

Other indicators of biodiversity would likely emerge in more extensive analyses. For example, if habitat for all native terrestrial vertebrates were modeled, as in gap analysis (Scott et al. 1993), other habitats would undoubtedly be identified. It is also possible that subsets of mature/overmature forest or riparian habitats would be targeted. For example, Knopf and Samson (1994) describe distinct differences in avian communities between lower- and upper-elevation riparian habitats within a drainage; in such instances, perhaps one habitat might be labeled an indicator of biodiversity and the other not. Ultimately, though, the presence of landscape indicators of biodiversity will depend upon the scale of the analysis, including resolution of data layers and detail of related attributes. What appears to be significant at one scale may not be apparent at others (Meentemeyer and Box 1987). However, extrapolating from this study, it seems fairly likely that late seral forests and riparian habitats will be accurate indicators of biodiversity throughout northwestern Montana.

### A Process for Evaluating Biodiversity

Finally, I would like to outline a generic process for evaluating past and present biological diversity at the landscape level, with a few comments on the resources necessary to complete such an assessment. Although no approach can be truly comprehensive -- every form of scientific investigation ultimately generates as many questions as answers -- this process involves a thorough examination of both rare and common elements in a landscape.

**1.** ***Define the extent of the study area.*** To a large degree, the scale of the analysis will determine the outcome; thus, selecting a specific area is a critical step. Regional context of the study area is also an important consideration; the area should not be an anomaly, but should represent broad-scale patterns across the surrounding region. Data resolution also plays a major part in determining results, and thus must also be selected with care.

2. ***Determine the goals and objectives of the analysis.*** Carefully framing the goals and objectives for conducting the analysis will help determine what data layers will be necessary, what analysis techniques should be employed, and what resources (time, people, and equipment) will be required. Computer software and hardware are an important consideration; the most powerful systems (like ARC/INFO on UNIX workstations) may have the steepest learning curves -- and the steepest prices. Obviously, trained personnel will be able to obtain faster results, and will be more likely to avoid common pitfalls. Close cooperation between a group of biologists and computer analysts (analogous to a Forest Service interdisciplinary team) would be most effective. Ideally, the study should be framed within a hierarchical context, so that inferences from broader-scale analyses may be applied to the study area, and in turn inferences for the study area may be applied to individual sites within the area.

3. ***Prepare the GIS database.*** Unless a complete GIS database exists for the study area, by far the largest investment of time and other resources will be expended at this stage. It is, however, a worthy investment, for once data layers are constructed, they can be readily updated and applied to many sorts of analyses. In addition, the famed "Garbage In, Garbage Out" principle comes into play at this point, because the accuracy of the data layers will place limits on the utility of the evaluation. Base layers important for an assessment of biodiversity include: vegetation (past and present), topography, hydrography, ownership and management, and roads. Other layers may be added if currently available or deemed necessary. Until they are examined carefully with project goals in mind, limited confidence should be placed in existing data layers, which may lack essential attributes, have limited locational accuracy, or have been prepared at coarser scale than needed. A generous portion of time should be allotted to modifying existing data layers to fit them to the desired analyses. Obviously, a

project able to incorporate more and better existing data will produce faster and better results.

**4.** ***Describe vegetative patterns and processes for the past and present landscape.*** Draw comparisons between current and historic vegetation, incorporating information for as many points in time as are available and deemed useful. To avoid "snapshot" comparisons, this step may be improved by modeling designed to assess a range of natural conditions in the presettlement landscape. In drawing comparisons between time periods, generalizations of vegetative type will probably be necessary; however, the loss of detail should be balanced by a corresponding gain in overall understanding of landscape patterns and function.

**5.** ***Identify wildlife species meriting special consideration in the study area, and model habitat and species distributions.*** Rare and endemic species, wide-ranging species, and species known or suspected to be sensitive to habitat alteration are all candidates for evaluation. In addition, efforts should be made to represent the spectrum of taxa found within the landscape of interest. Gather information on habitat selection and distributions from existing literature, Natural Heritage Database records, and agency records, among other sources. Prepare models of habitat and have them reviewed by biologists most familiar with individual species. Models should be prepared for multiple time periods to approximate trends in habitat and thus population status.

**6.** ***Identify other elements worthy of consideration in a fine-filter approach.*** Acquiring locational information for rare plants, animals, plant communities, geologic features, and other unique habitats is another critical aspect of an evaluation of biodiversity. The Natural Heritage Database, agency records, and existing literature should be reviewed to identify such locations. Survey effort must be evaluated at this stage; not all areas have been equally well surveyed, and thus bias may be introduced.

**7.** ***Assess protection of biodiversity within each ownership/management zone.*** Select a scheme for assessing protection of biodiversity under various ownership and management regimes; one example is the classification system drafted by the Montana Interagency Natural Areas Committee (1993). Assign codes for protection status to each ownership/management zone, and describe the existing network of protected areas as well as the surrounding landscape.

**8.** ***Assess representation of desired landscape features within the existing network.*** Representation of cover types and biophysical zones may be most important, because taken together, they represent existing and potential vegetative patterns within a landscape. However, the most straightforward and useful assessment may be representation of elevation zones; although it is well-recognized that high elevation areas are disproportionately represented in reserves throughout the western United States, a simple graphical illustration of this phenomenon can be very effective. Representation of rare elements (step 6) and wildlife habitat (as modeled in step 5) should also be evaluated.

**9.** ***Identify desired features for additional reserves.*** Examples include underrepresented cover types or elevation zones, mature/overmature forest, low road densities, and presence of rare plant species. Create a database with attributes indicating the presence or absence of desired characteristics. Explore various scoring methods to identify the highest-priority sites for supplementation of the existing reserve network. Iterative algorithms may also be employed to select sites in an efficient and repeatable manner (Margules et al. 1988, Margules 1989, Pressey and Nicholls 1989, Pressey et al. 1993).

**10.** ***Identify target sites and their present ownership/management status.*** To refine the above set, select the largest examples in the best locations as priorities for acquisition or changes in management direction. If landscape connectivity is poor, potential linkage areas should also be targeted. Work cooperatively with the landowners involved to secure sites of the highest priority.

**11.** ***Design a network of protected areas.*** For the sites selected above, delineate boundaries, giving preference to topographic breaks and other meaningful distinctions over administrative boundaries. To minimize edge effects, protected areas should be nearly circular; shapes with high perimeter/area ratios should be avoided. Where possible, provide buffer zones to mitigate outside influences and linkage areas to maintain connections between protected areas. In the design process, consideration of natural landscape patterns and processes is essential.

**12.** ***Evaluate results in terms of goals and objectives.*** As is always the case in dealing with natural systems, our efforts to manage protected areas are directed toward a moving target which will respond to the changes we make (May 1994), and not always in a predictable manner. At this stage in the evaluation process, highlight further needs, which at a minimum should include field validation of potential reserves. Validation of data layers, especially wildlife models, would also be valuable. A final, critical step involves interpreting the results of the evaluation in relation to overall land management in the matrix surrounding the network of protected areas, for it is in the managed matrix that efforts to conserve biodiversity will ultimately succeed or fail (Franklin 1993).

## SUMMARY

Since the 1930s, the Seeley-Swan landscape has become increasingly fragmented, and proportions of individual cover types have shifted as timber harvest has replaced fire as the dominant disturbance process. In particular, mature/overmature forests, the landscape's matrix component in the 1930s, have declined in total area, while seedling and sapling seral stages have become more extensive and could potentially replace mature/overmature forests as the landscape matrix. This shift is reflected in habitat predictions for wildlife species using older forests; in general, habitat has declined in total area and become more fragmented in its configuration. Although a substantial proportion of the landscape is already accorded high protection, the lower elevations and associated cover types and biophysical zones are poorly represented in the existing reserve network. Inclusion of low-elevation old-growth forests -- particularly ponderosa pine, western red cedar, and extensive stands of mixed conifer composition such as those blanketing the valley floor in the 1930s -- would improve the existing reserve network. In addition, the Swan Valley provides habitat for numerous sensitive plant species. Small reserves have been proposed to protect these rare plants; these areas could be expanded to provide a buffer around plant locations, minimize outside influences, and increase the probability of these reserves playing a functional role in maintaining healthy ecosystem and landscape processes. The process of augmenting the existing network of protected areas will require intensive cooperative efforts because of the number of landowners involved; key players include the Flathead and Lolo National Forests, Plum Creek Timber Company, the Montana Department of State Lands, and many individual landowners. Successful coordination offers great rewards. Because of its unique elements, the Seeley-Swan landscape merits exceptional efforts toward conservation of biodiversity.

Table 5-1. Draft classification of management and protection levels, including hierarchical codes, for the state of Montana (Montana Interagency Natural Areas Committee 1993).

| | OWNERSHIP | | PROTECTION | | DESIGNATION | EXAMPLES |
|---|---|---|---|---|---|---|
| 1 | Public | 11 | High | 111 | Strong | Wilderness Areas, Biosphere Reserves, National Parks |
| | | | | 112 | Moderate | Primitive Areas, Outstanding Natural Areas |
| | | 12 | Moderate | 121 | Strong | National Wildlife Refuges, National Recreation Areas, Wildlife Management Areas, State Parks |
| | | | | 122 | Moderate | grizzly bear habitat, old growth, riparian areas |
| | | 13 | Minimum | 131 | Not managed for biodiversity values | timber and grazing lands |
| | | | | 132 | Concentrated development/use | mining sites, campgrounds, ski resorts |
| 2 | Private | 21 | Protected | 210 | Formally designated nature preserves | The Nature Conservancy and National Audubon Society preserves |
| | | 22 | Semi-protected | 220 | Certain natural features protected | conservation easements, registry |
| | | 23 | Unprotected | 230 | No formally designated protection | |

Table 5-2. National Forest management areas in the Seeley-Swan landscape (USDA:FS 1985, 1986) and protection codes (Montana Interagency Natural Areas Committee 1993; see Table 5-1) assigned for this study.

| Management | Designation and General Management Objectives | Protection |
|---|---|---|
| | **Flathead National Forest** | |
| 1 | Unsuitable for timber harvest, maintain present conditions | 122 |
| 2 | Primitive Recreation Opportunity Spectrum (ROS), unroaded | 122 |
| 2a | Semiprimitive nonmotorized ROS, unroaded | 122 |
| 2b | Semiprimitive motorized ROS, unroaded | 122 |
| 2e | Candidate Research Natural Area (semiprimitive nonmotorized) | 111 |
| 5 | Timberlands, high scenic value -- retention visual quality objective | 122 |
| 7 | Timberlands, high scenic value -- partial retention VQO | 131 |
| 9 | White-tailed deer winter habitat -- suitable for timber harvest | 131 |
| 10 | Administrative sites | 132 |
| 11c | Grizzly bear travel corridor (Clearwater Divide) -- suitable timber | 131 |
| 12 | Riparian areas -- unsuitable for timber harvest | 122 |
| 12a | Swan River Island Research Natural Area | 111 |
| 13 | Mule deer and elk winter habitat -- suitable for timber harvest | 131 |
| 15 | Suitable timber lands | 131 |
| 15c | White-tailed deer summer habitat -- suitable for timber harvest | 131 |
| 16 | Suitable timber lands -- aerial logging | 131 |
| 17 | Riparian areas -- suitable for timber harvest (long rotation) | 131 |
| 22 | Wilderness (Mission Mountains) | 111 |
| | **Lolo National Forest** | |
| 1 | Unsuitable for timber harvest, maintain near-natural conditions | 122 |
| 2 | Administrative sites | 132 |
| 6 | Proposed Research Natural Areas | 111 |
| 7 | Campgrounds and picnic areas | 132 |
| 9 | Concentrated public use | 132 |
| 11 | Unsuitable for timber, large roadless areas, old-growth wildlife | 122 |
| 12 | Existing/proposed wilderness | 111 |
| 13 | Lakes/riparian areas, some suitable for timber harvest, some not | 122 |
| 16 | Suitable timber lands | 131 |
| 17 | Suitable timber lands -- mostly >60% slope | 131 |
| 20 | Essential grizzly bear habitat -- suitable for timber harvest | 131 |
| 20a | Essential grizzly bear habitat -- unsuitable for timber harvest | 122 |
| 24 | High visual sensitivity -- retention VQO | 122 |
| 25 | High visual sensitivity -- partial retention VQO | 131 |
| 26 | Elk summer habitat -- suitable for timber harvest | 131 |

Table 5-3. Biophysical zones modeled for the Seeley-Swan landscape based on 4 x 4 matrix of temperature and moisture (Fig. 5-1), assumed to represent aggregations of habitat types (Pfister et al. 1977).

| BIOPHYSICAL ZONE | DESCRIPTION AND REPRESENTATIVE HABITAT TYPES (Menakis, pers. comm.) |
|---|---|
| 1 | **MODERATELY (MOD) WET/COLD - HERBACEOUS** |
| 2 | **MOD WET/COLD - FORESTED**<br>*Abies lasiocarpa/Luzula hitchcockii*<br>*Abies lasiocarpa/Menziesia ferruginea*<br>*Larix lyallii - Abies lasiocarpa* |
| 3 | **MOD DRY/COLD - FORESTED**<br>*Abies lasiocarpa/Vaccinium scoparium*<br>*Abies lasiocarpa - Pinus albicaulis/Vaccinium scoparium* |
| 4 | **DRY/COLD - FORESTED**<br>*Pinus albicaulis - Abies lasiocarpa* |
| 6 | **MOD WET/MOD COLD - FORESTED**<br>*Abies lasiocarpa/Clintonia uniflora*<br>*Abies lasiocarpa/Linnaea borealis*<br>*Picea/Clintonia uniflora*<br>*Picea/Galium triflorum* |
| 7 | **MOD DRY/MOD COLD - FORESTED**<br>*Abies grandis/Xerophyllum tenax*<br>*Abies lasiocarpa/Vaccinium caespitosum*<br>*Abies lasiocarpa/Vaccinium globulare*<br>*Abies lasiocarpa/Xerophyllum tenax*<br>*Pseudotsuga menziesii/Linnaea borealis* |
| 10 | **MOD WET/MOD WARM - FORESTED**<br>*Abies grandis/Linnaea borealis*<br>*Abies grandis/Clintonia uniflora*<br>*Thuja plicata/Clintonia uniflora* |
| 11 | **MOD DRY/MOD WARM - FORESTED**<br>*Pseudotsuga menziesii/Calamagrostis rubescens*<br>*Pseudotsuga menziesii/Physocarpus malvaceus*<br>*Pseudotsuga menziesii/Vaccinium globulare* |
| 99 | **BARREN** |

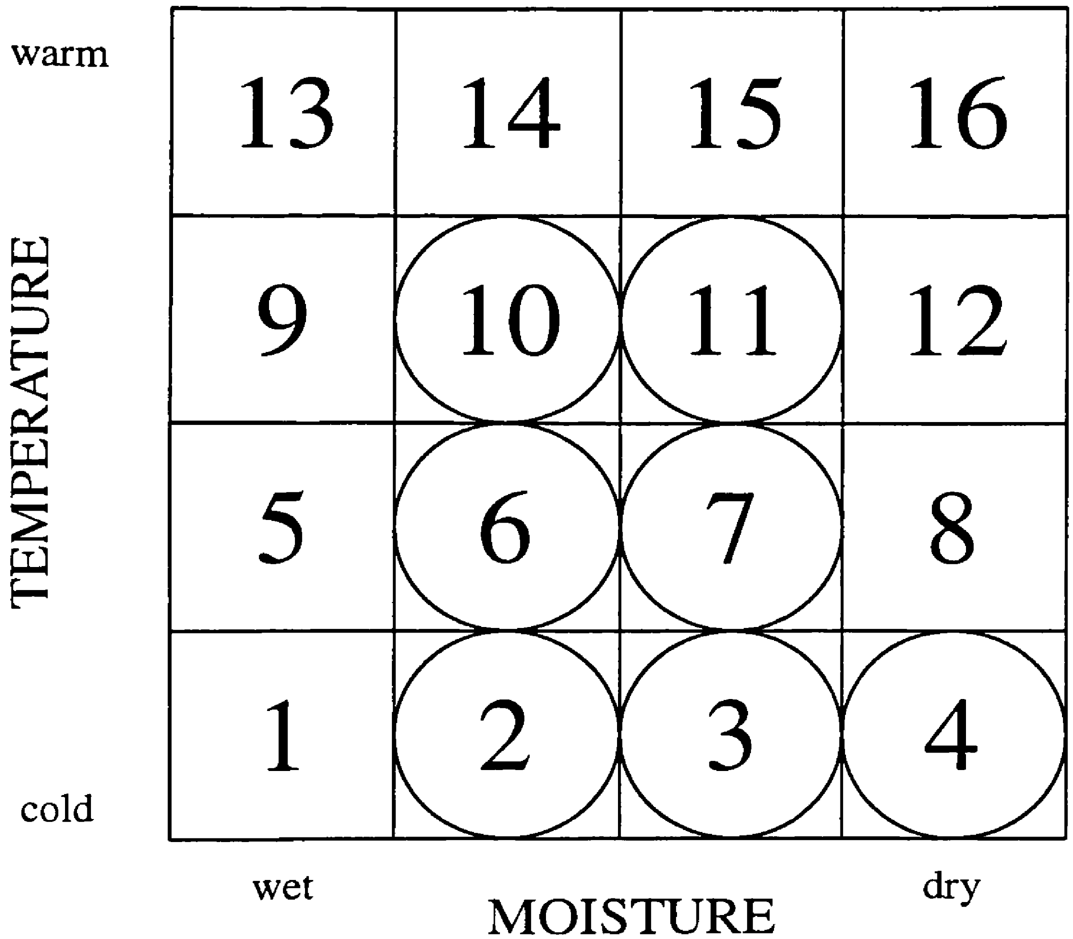


Figure 5-1. Four-by-four matrix of temperature and moisture regimes, assumed to represent aggregations of habitat types (Pfister et al. 1977), used in modeling biophysical zones. Circled types were modeled for the Seeley-Swan landscape of northwestern Montana.

Table 5-4. Modeling rules used to map biophysical zones in the Seeley-Swan landscape, northwestern Montana (Mantas and Sirucek, Flathead NF; Menakis, pers. comm.).

| BIOPHYSICAL ZONE[a] | ELEVATIONAL RANGE (ft.) | ASPECT[b] N and E | ASPECT[b] S and W | ASPECT[b] Flat | SLOPE (%) 5-30 | SLOPE (%) 30-60 | SLOPE (%) 60+ |
|---|---|---|---|---|---|---|---|
| North of Clearwater Divide: | | | | | | | |
| 1 | 7500-8500+ | X | X | X | X | X | |
| 2 | 5500-6500 | X | | X | X | X | X |
| 3 | 6500-7000 | X | | | X | X | X |
| 4 | 6500-7500 | | X | X | X | X | X |
| | 7000-7500 | X | | | X | X | X |
| 6 | 2500-3500 | X | X | X | X | X | X |
| | 4500-5500 | X | X | X | X | X | X |
| 7 | 5500-6500 | | X | | X | X | X |
| 10 | 3500-4500 | X | X | X | X | X | X |
| 99 | 7500+ | X | X | | | | X |
| South of Clearwater Divide: | | | | | | | |
| 1 | 7500-8500+ | X | X | X | X | X | |
| 2 | 5000-6500 | X | | X | X | X | X |
| 3 | 6500-7000 | X | | | X | X | X |
| 4 | 6500-7500 | | X | X | X | X | X |
| | 7000-7500 | X | | | X | X | X |
| 6 | 2500-4000 | X | X | X | X | X | X |
| | 4000-5000 | X | | | | X | X |
| 7 | 5500-6500 | | X | | X | X | X |
| 10 | 4000-5000 | X | | X | X | | |
| 11 | 4000-5500 | | X | | X | X | X |
| 99 | 7500+ | X | X | | | | X |

[a] For descriptions, see Table 5-3.
[b] N and E = 315-360° and 0-135°; S and W = all other aspects; Flat = <5% slope.

Table 5-5. Sensitive plant species found in the Seeley-Swan landscape, grouped according to habitat.

| PLANT GROUP | HABITAT | SPECIES |
|---|---|---|
| 1 | Aquatic | *Howellia aquatilis* |
| 2 | Aquatic | *Bidens beckii* |
| | | *Brasenia schreberi* |
| | | *Potamogeton obtusifolius* |
| | | *Scirpus subterminalis* |
| | | *Utricularia intermedia* |
| 3 | Fen and other riparian | *Carex livida* |
| | | *Carex paupercula* |
| | | *Cypripedium calceolus* |
| | | *Cypripedium passerinum* |
| | | *Drosera anglica* |
| | | *Dryopteris cristata* |
| | | *Eleocharis rostellata* |
| | | *Epipactis gigantea* |
| | | *Eriophorum viridicarinatum* |
| | | *Liparis loeselii* |
| | | *Lycopodium inundatum* |
| | | *Ophioglossum vulgatum* |
| | | *Viola renifolia* |
| 4 | Forest and nonriparian forest openings | *Allium fibrillum* |
| | | *Botrychium montanum* |
| | | *Botrychium spathulatum* |
| | | *Cypripedium fasciculatum* |
| | | *Gaultheria ovatifolia* |
| | | *Grindelia howellii* |
| | | *Madia minima* |
| 5 | Alpine and subalpine | *Cardamine rupicola* |
| | | *Polystichum kruckebergii* |
| | | *Synthyris canbyi* |

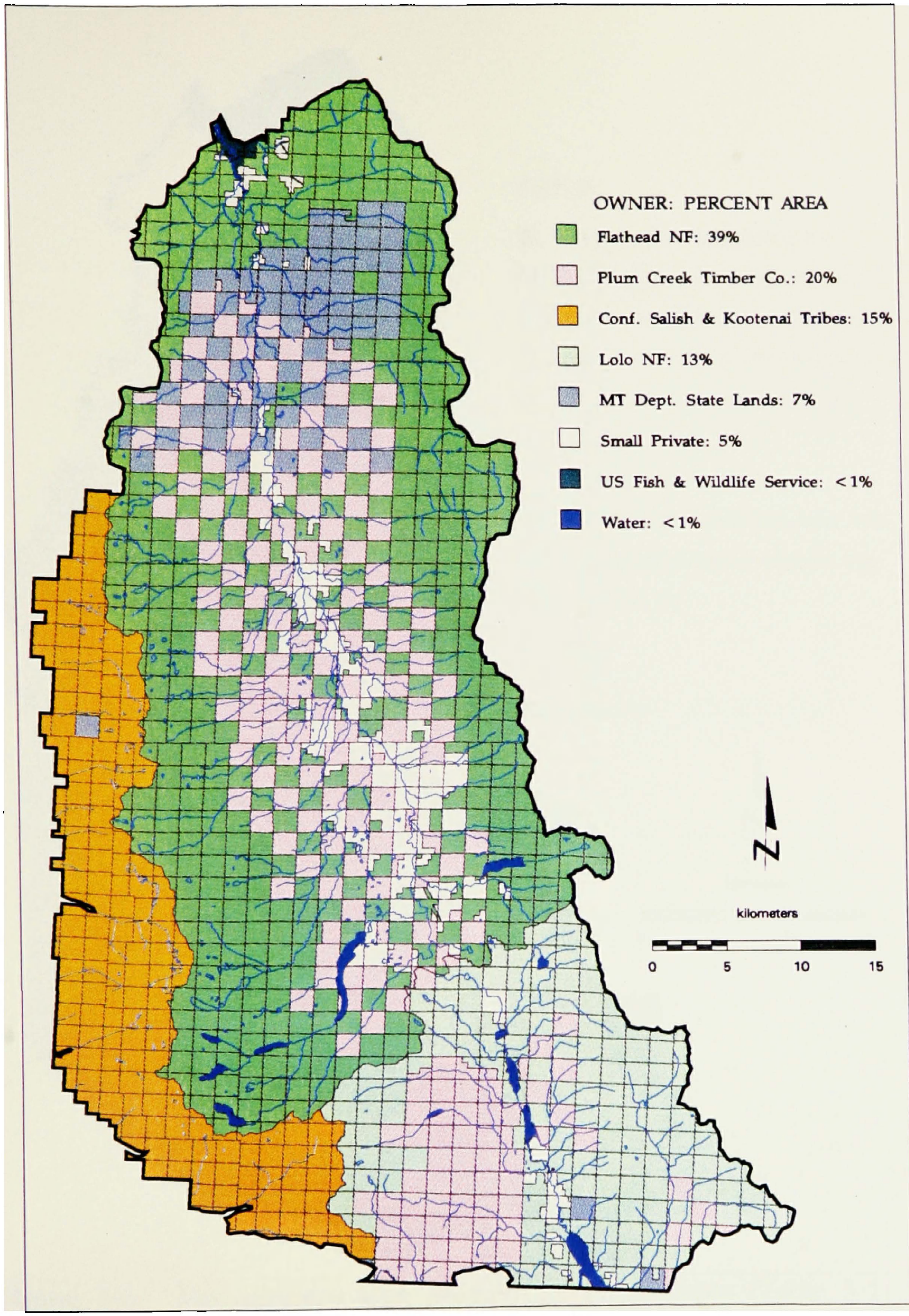


Figure 5-2. Land ownership patterns in the Seeley-Swan landscape, northwestern Montana.

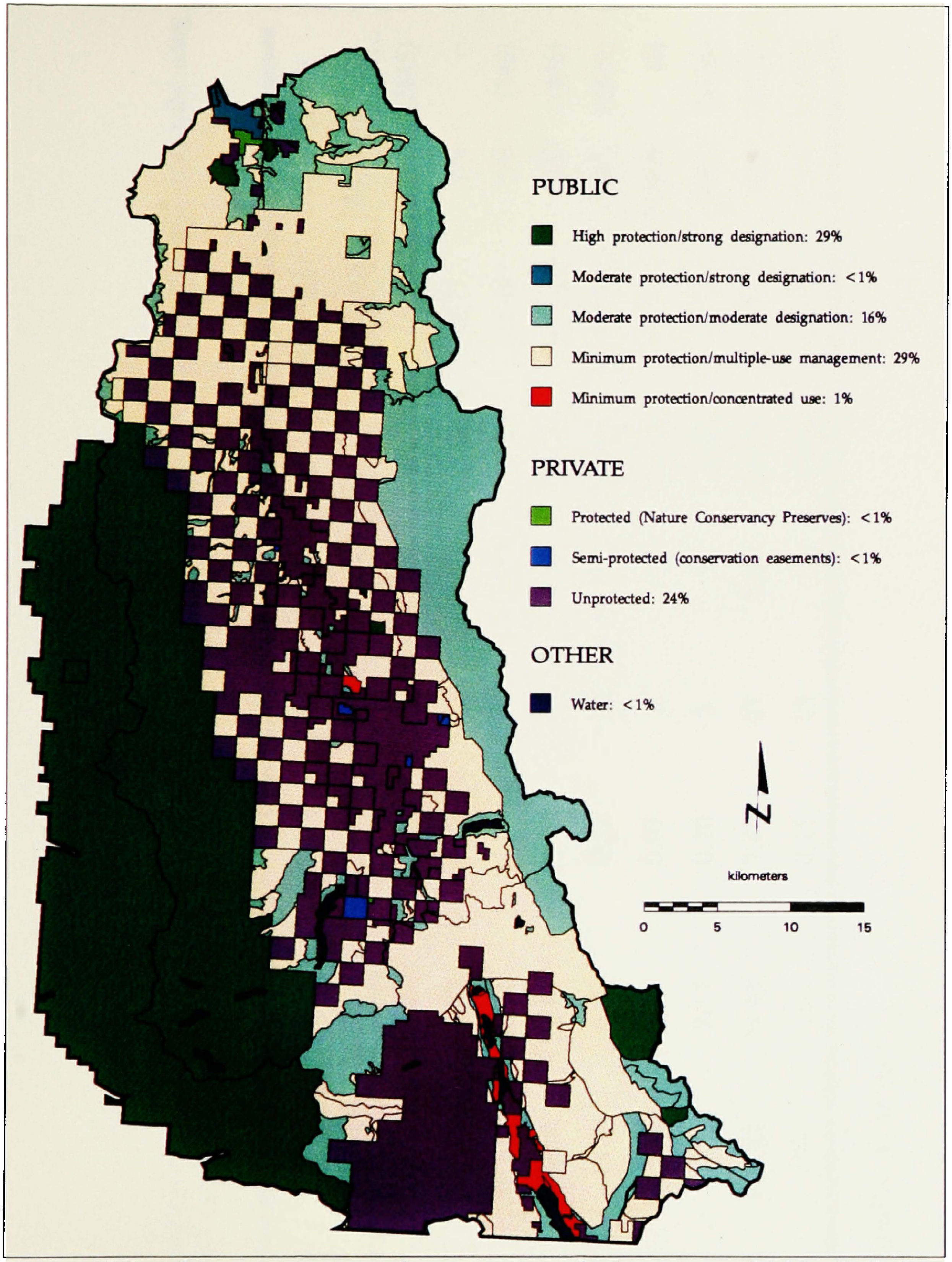


Figure 5-3. Management and protection designations (Table 5-1) in the Seeley-Swan landscape, northwestern Montana. Lines indicate management area boundaries; some adjacent areas are assigned the same code.

Table 5-6. Spatial statistics for each protection level in the Seeley-Swan landscape, northwestern Montana, calculated using FRAGSTATS (McGarigal and Marks 1994).

| Protection Level | Hectares | Percent Landscape | Largest Patch Index | Number Patches | Mean Patch Size (SD)[a] | | Mean Core Area Index | Mean Nearest Neighbor (SD) | |
|---|---|---|---|---|---|---|---|---|---|
| 111 | 70,730 | 28.53 | 27.52 | 8 | 8841 | (22,451) | 77.67 | 3730 | (2845) |
| 121 | 683 | 0.28 | 0.28 | 1 | 683 | (0) | 84.71 | n/a | -- |
| 122 | 40,680 | 16.41 | 11.54 | 77 | 528 | (3250) | 49.14 | 710 | (740) |
| 131 | 71,617 | 28.89 | 13.48 | 46 | 1557 | (5803) | 64.11 | 392 | (846) |
| 132 | 1371 | 0.55 | 0.22 | 10 | 137 | (174) | 51.70 | 5841 | (8529) |
| 210 | 181 | 0.07 | 0.07 | 2 | 91 | (74) | 61.79 | 55,058 | (0) |
| 220 | 427 | 0.17 | 0.10 | 5 | 85 | (88) | 65.15 | 6461 | (3706) |
| 230 | 60,682 | 24.48 | 17.40 | 27 | 2247 | (8481) | 64.83 | 605 | (517) |
| water | 1554 | 0.63 | 0.15 | 13 | 120 | (108) | 63.45 | 2753 | (2190) |

[a] SD = standard deviation

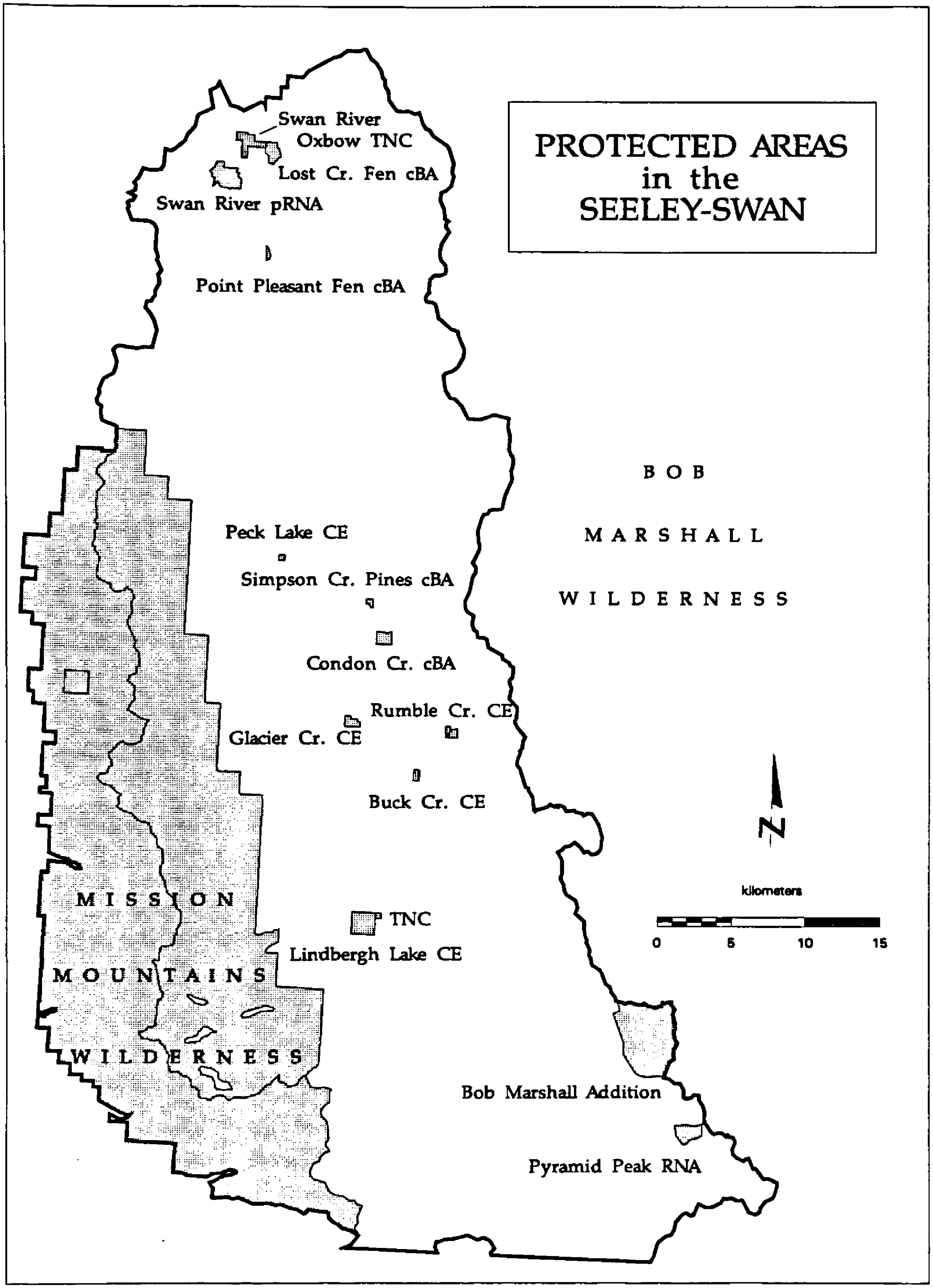


Figure 5-4. Existing or proposed protected areas in the Seeley-Swan landscape, including wilderness, research natural areas (RNA), candidate botanical areas (cBA), Nature Conservancy preserves (TNC), and conservation easements (CE).

Table 5-7. Existing and proposed protected areas in the Seeley-Swan landscape, northwestern Montana.

| SITE | HECTARES | OWNER (HELD BY)[a] |
|---|---|---|
| **Level 111 Protection Status:** | | |
| Swan River Research Natural Area (proposed) | 276 | Flathead NF |
| Pyramid Peak Research Natural Area | 210 | Lolo NF |
| Condon Creek Botanical Area (proposed) | 93 | Flathead NF |
| Lost Creek Fen Botanical Area (candidate) | 101 | Flathead NF |
| Simpson Creek Pines Botanical Area (candidate) | 40 | Flathead NF |
| Point Pleasant Fen Botanical Area (candidate) | 20 | Montana Dept. State Lands |
| Bob Marshall Wilderness Addition | 1770 | Lolo NF |
| Mission Mountains Wilderness | 68,570 | Flathead NF, Confederated Salish & Kootenai Tribes |
| **Level 210 Protection Status:** | | |
| Swan River Oxbow Preserve | 165 | The Nature Conservancy |
| Preserve adjacent to Lindbergh Lake CE | 16 | The Nature Conservancy |
| **Level 220 Protection Status:** | | |
| Peck Lake Conservation Easement | 16 | Flathead NF |
| Glacier Creek Conservation Easement | 65 | Montana Land Reliance |
| Rumble Creek Conservation Easement No. 1 | 19 | Montana Land Reliance |
| Rumble Creek Conservation Easement No. 2 | 36 | The Nature Conservancy |
| Buck Creek Conservation Easement | 32 | Institute of the Rockies |
| Lindbergh Lake Conservation Easement | 259 | The Nature Conservancy |

[a] Ownership listed for all sites except conservation easements, where the entity holding the easement is listed.

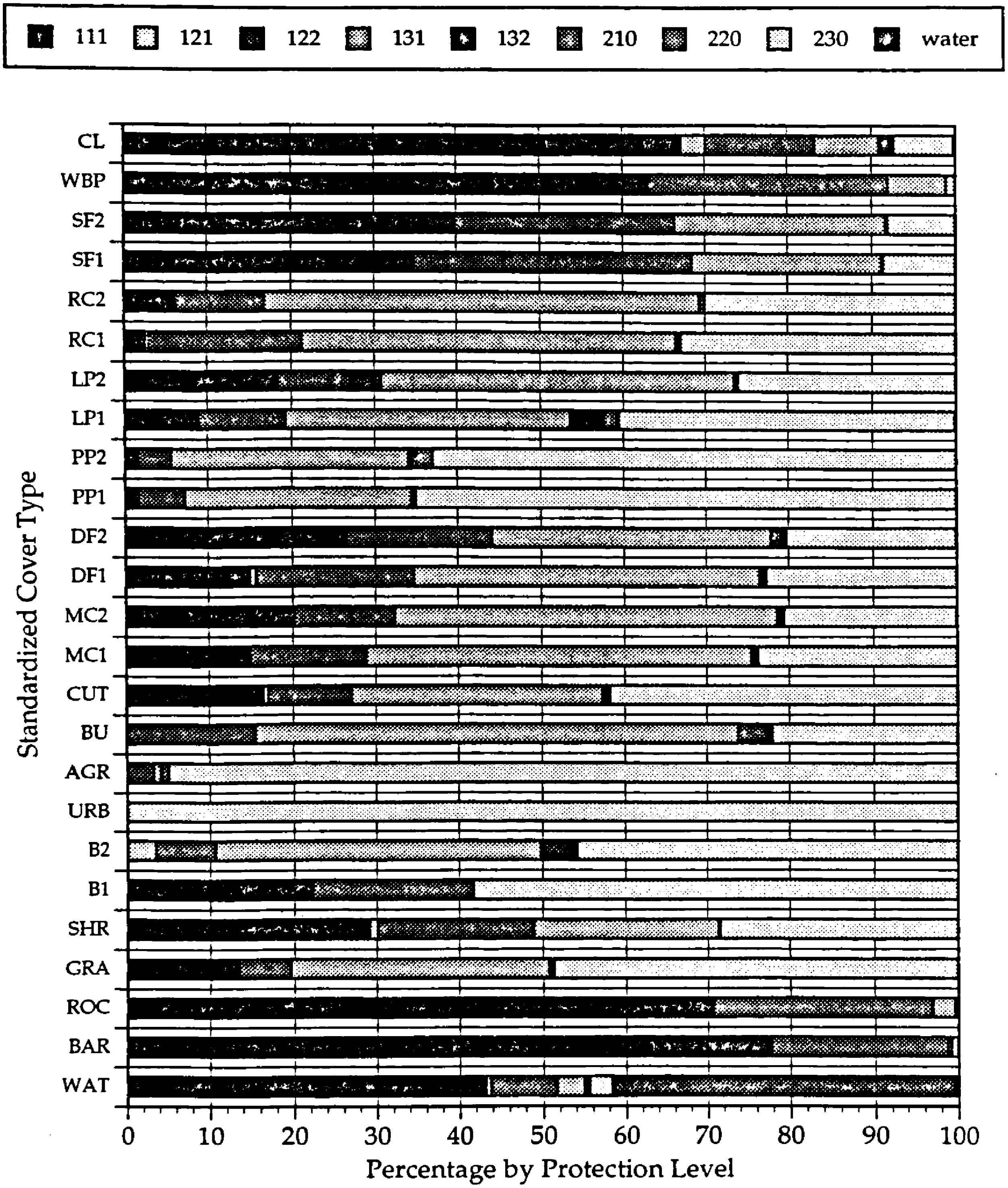


Figure 5-5. Proportion of each cover type by protection level, Seeley-Swan landscape, northwestern Montana. Protection levels increase sequentially from the left, as shown for the legend.

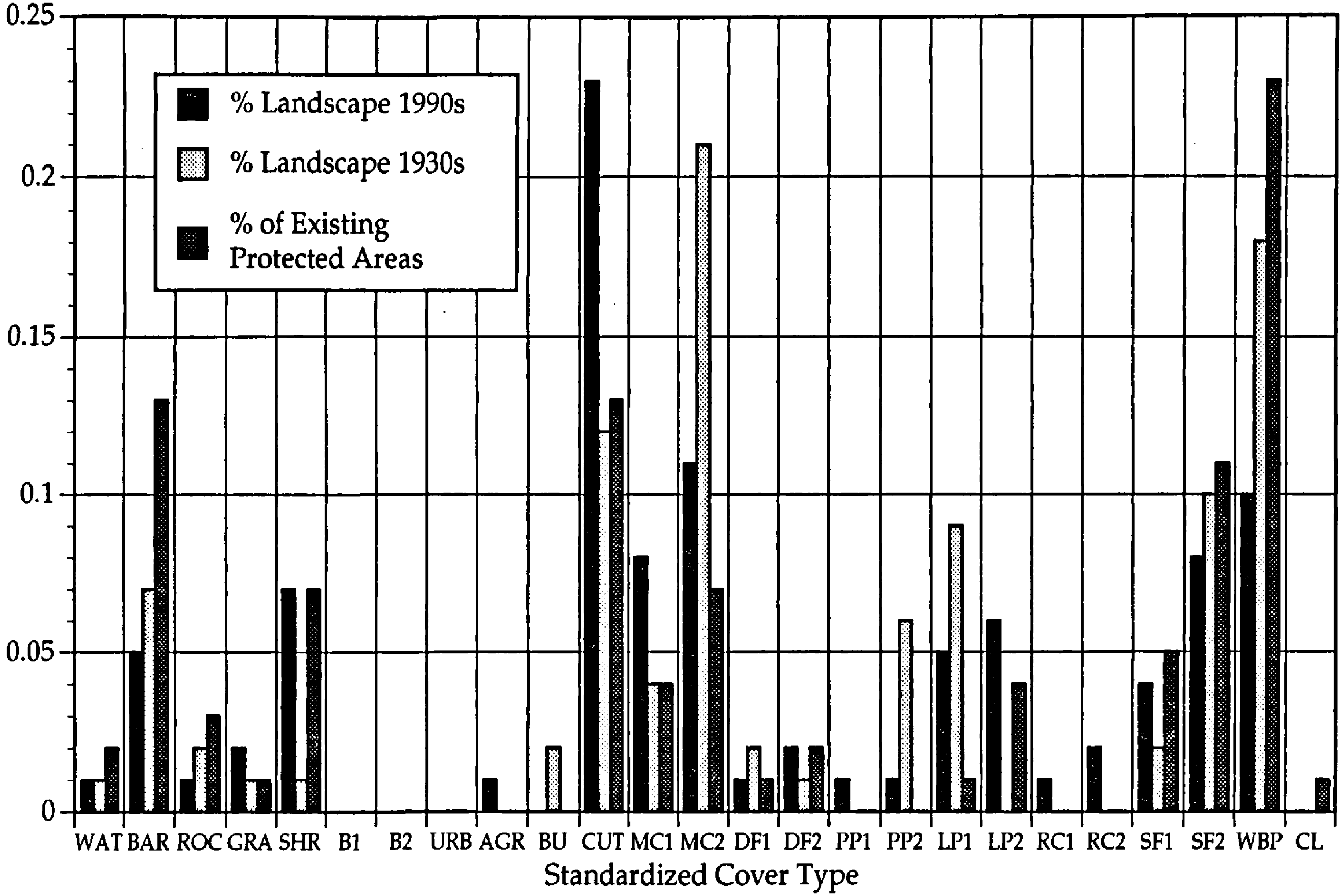


Figure 5-6. Representation of standardized cover types in existing protected areas (Levels 111 and 210) in relation to the proportion of each cover type in the past and present Seeley-Swan landscape, northwestern Montana.

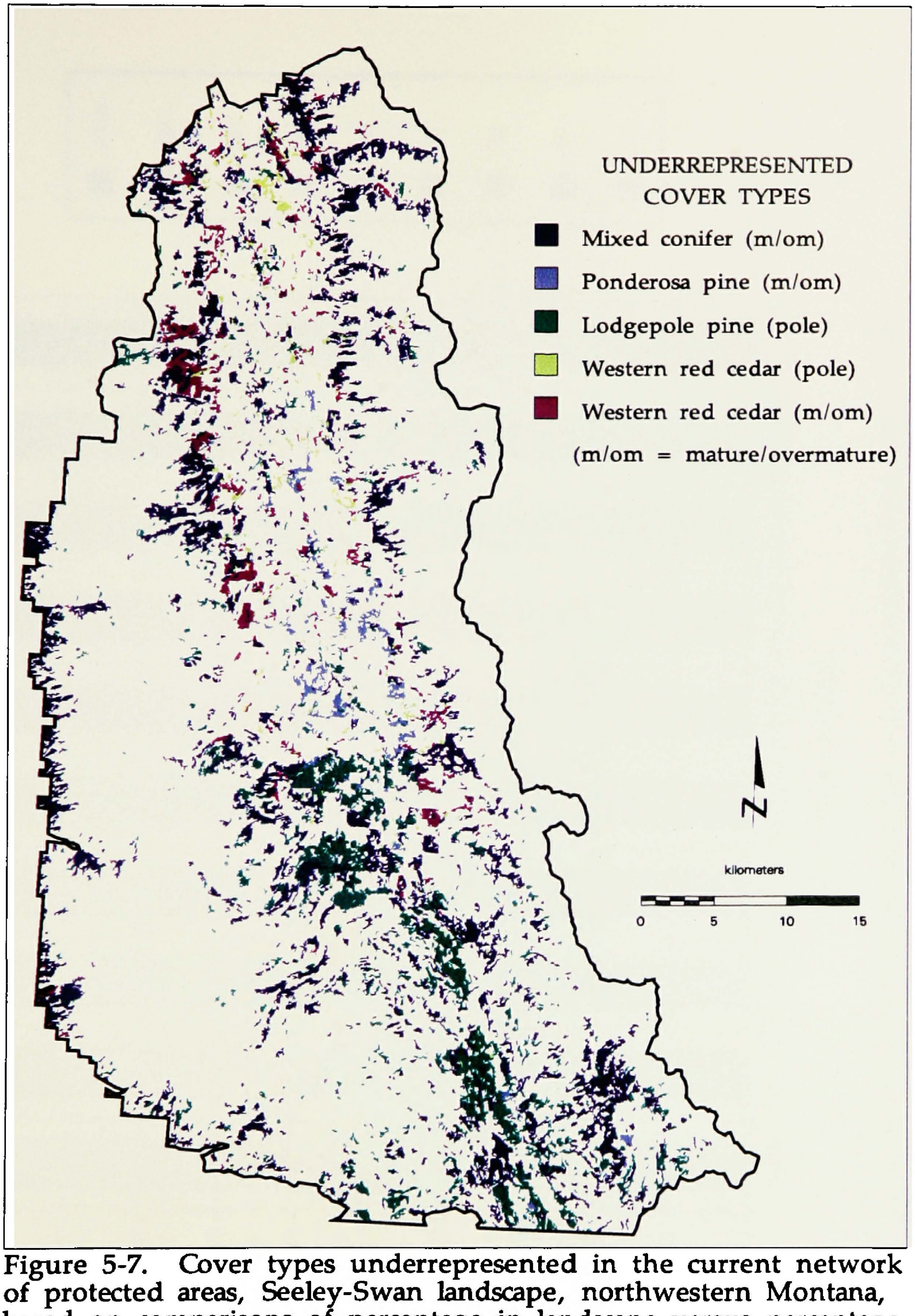


Figure 5-7. Cover types underrepresented in the current network of protected areas, Seeley-Swan landscape, northwestern Montana, based on comparisons of percentage in landscape versus percentage in existing protected areas.

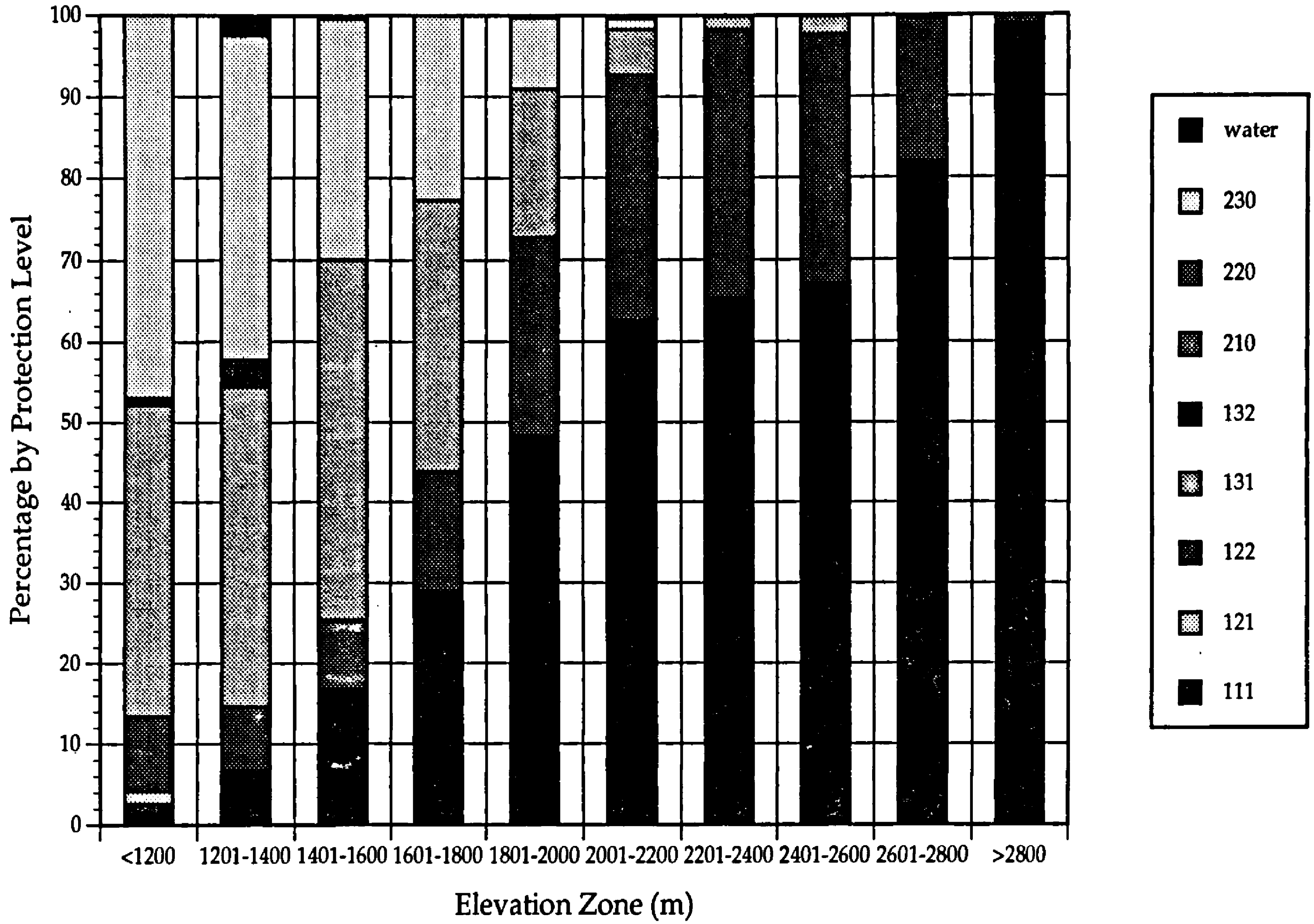


Figure 5-8. Proportion of each elevation zone by protection level, Seeley-Swan landscape, northwestern Montana. Protection levels increase sequentially from the origin, as shown for the legend.

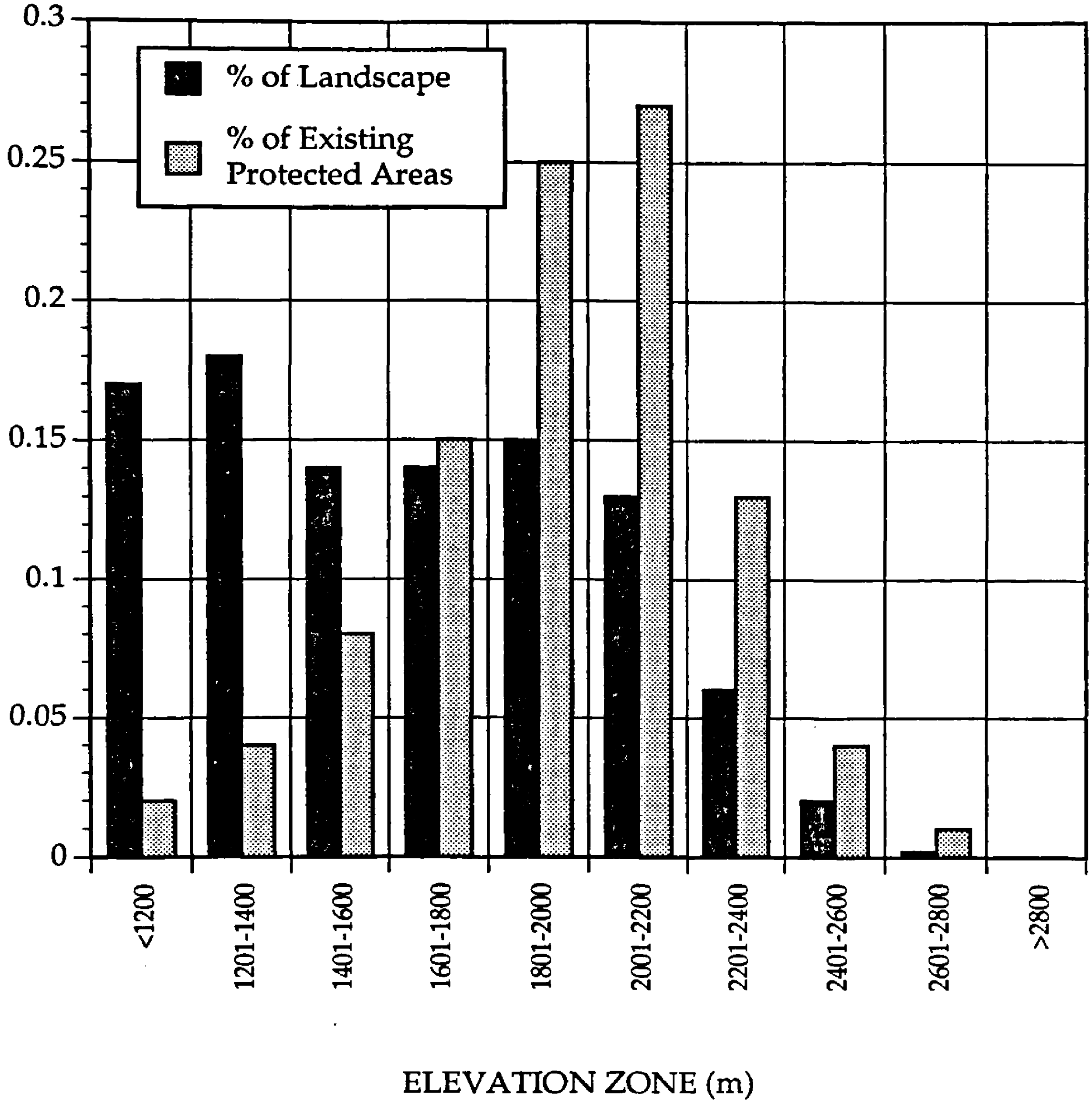


Figure 5-9. Representation of 200 m elevation zones within existing protected areas (Levels 111 and 210) in relation to the proportion of the landscape within each elevation zone, Seeley-Swan landscape, northwestern Montana.

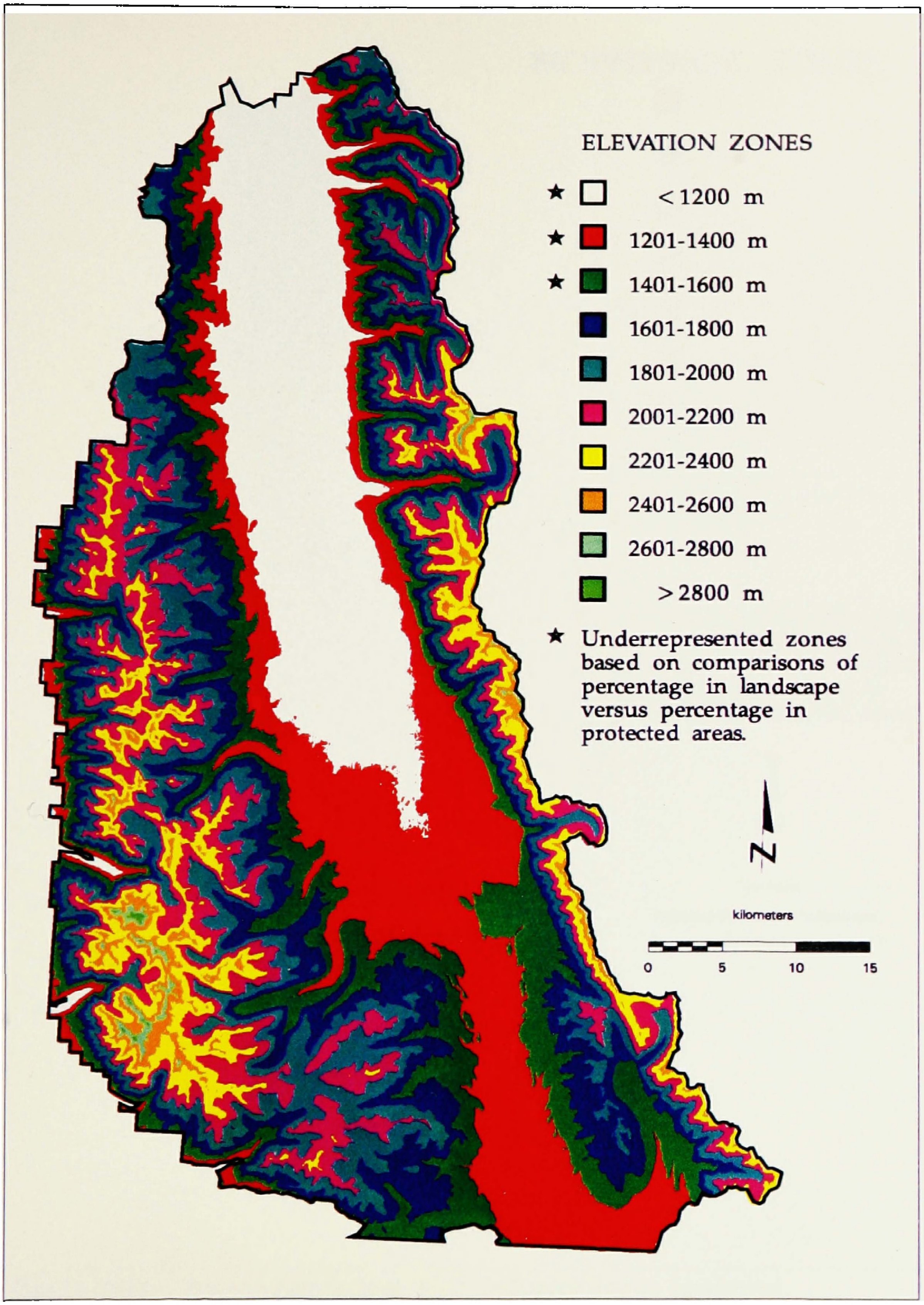


Figure 5-10. Elevation zones (200 m intervals) in the Seeley-Swan landscape, northwestern Montana.

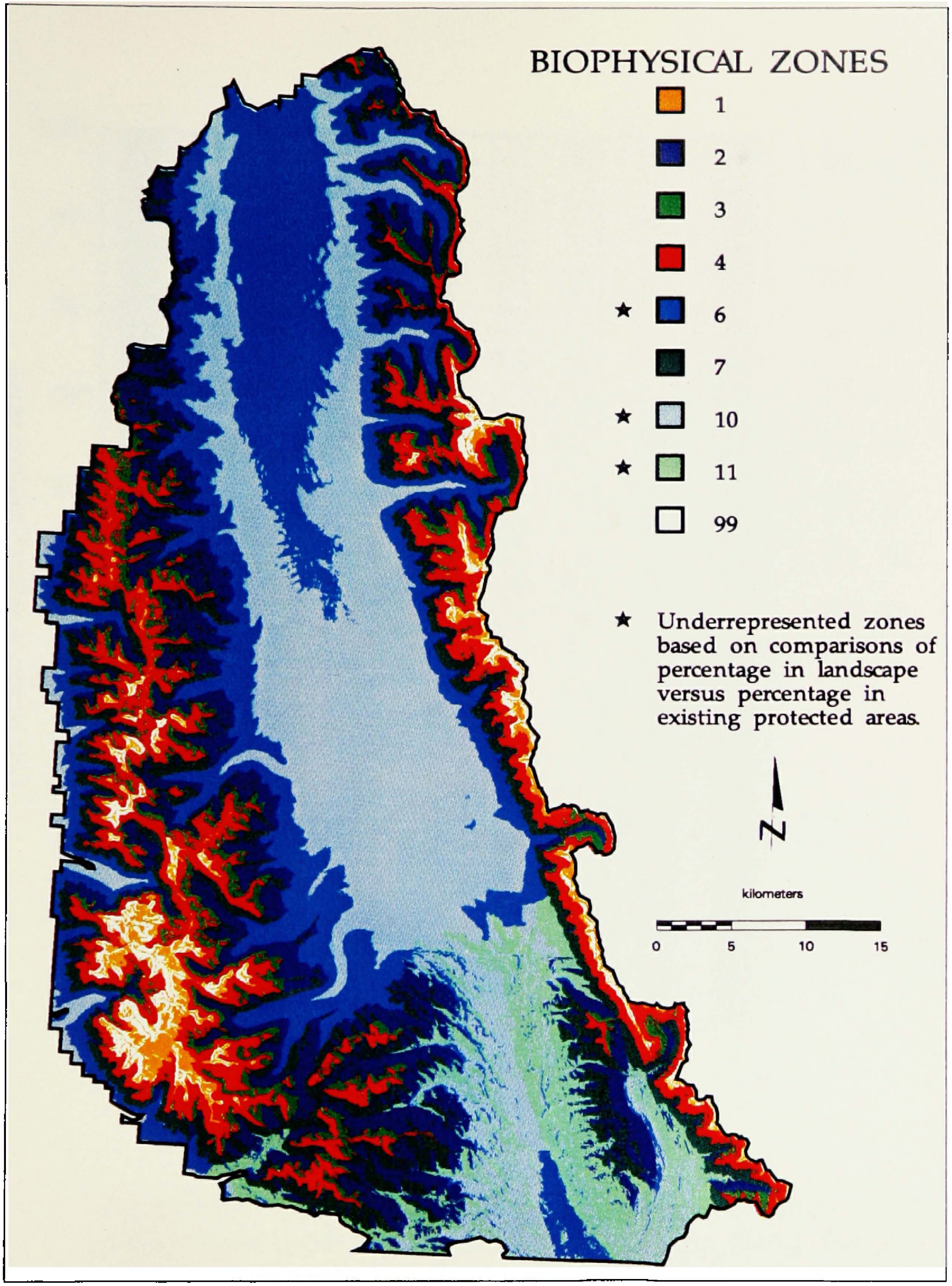


Figure 5-11. Biophysical zones in the Seeley-Swan landscape, northwestern Montana. Model based on a 4 x 4 matrix of temperature and moisture regimes assumed to represent aggregations of habitat types (Pfister et al. 1977). See Table 5-3 for description of zones.

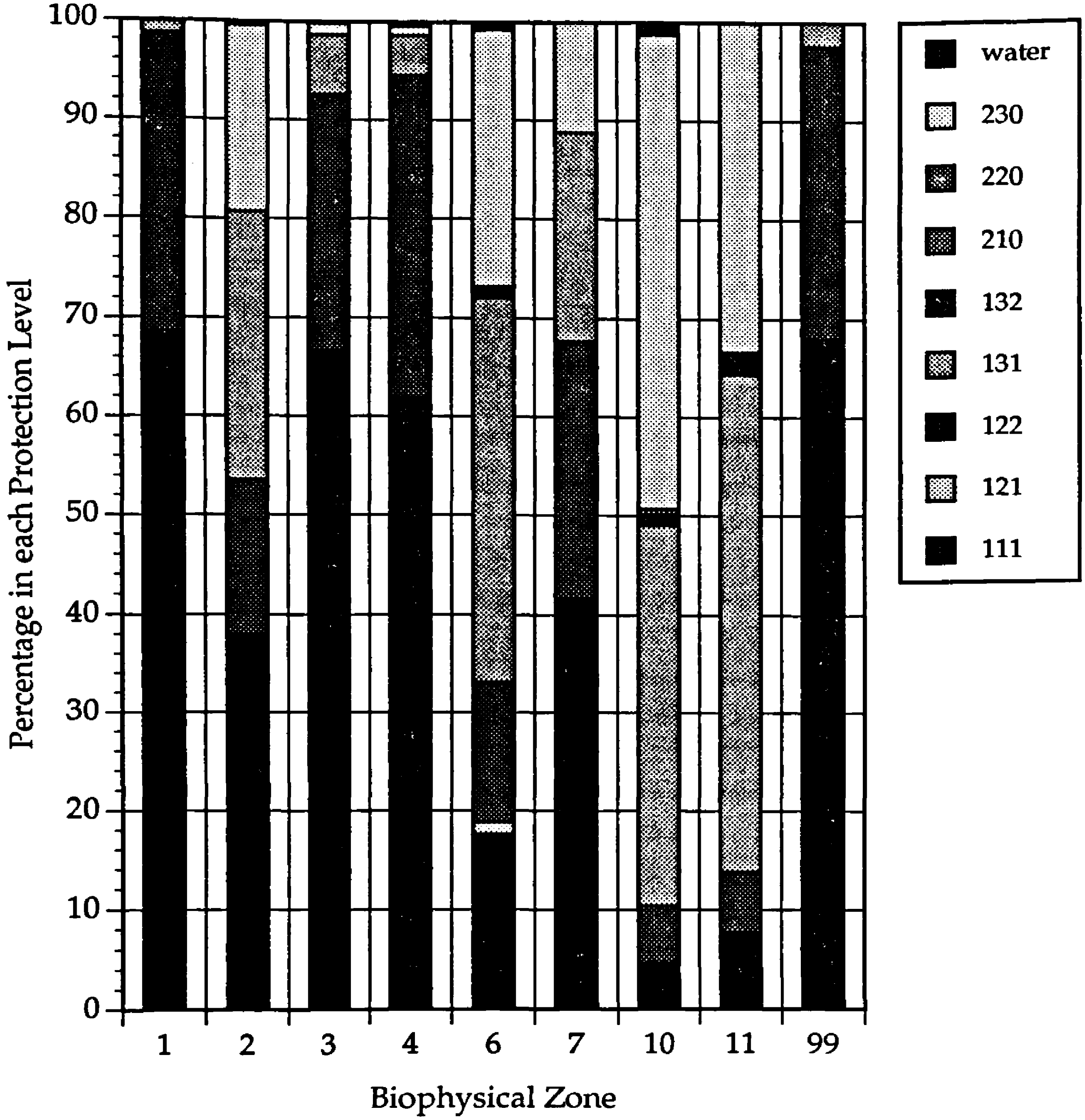


Figure 5-12. Proportion of each biophysical zone by protection level, Seeley-Swan landscape, northwestern Montana. Protection levels increase sequentially from the origin, as shown for the legend.

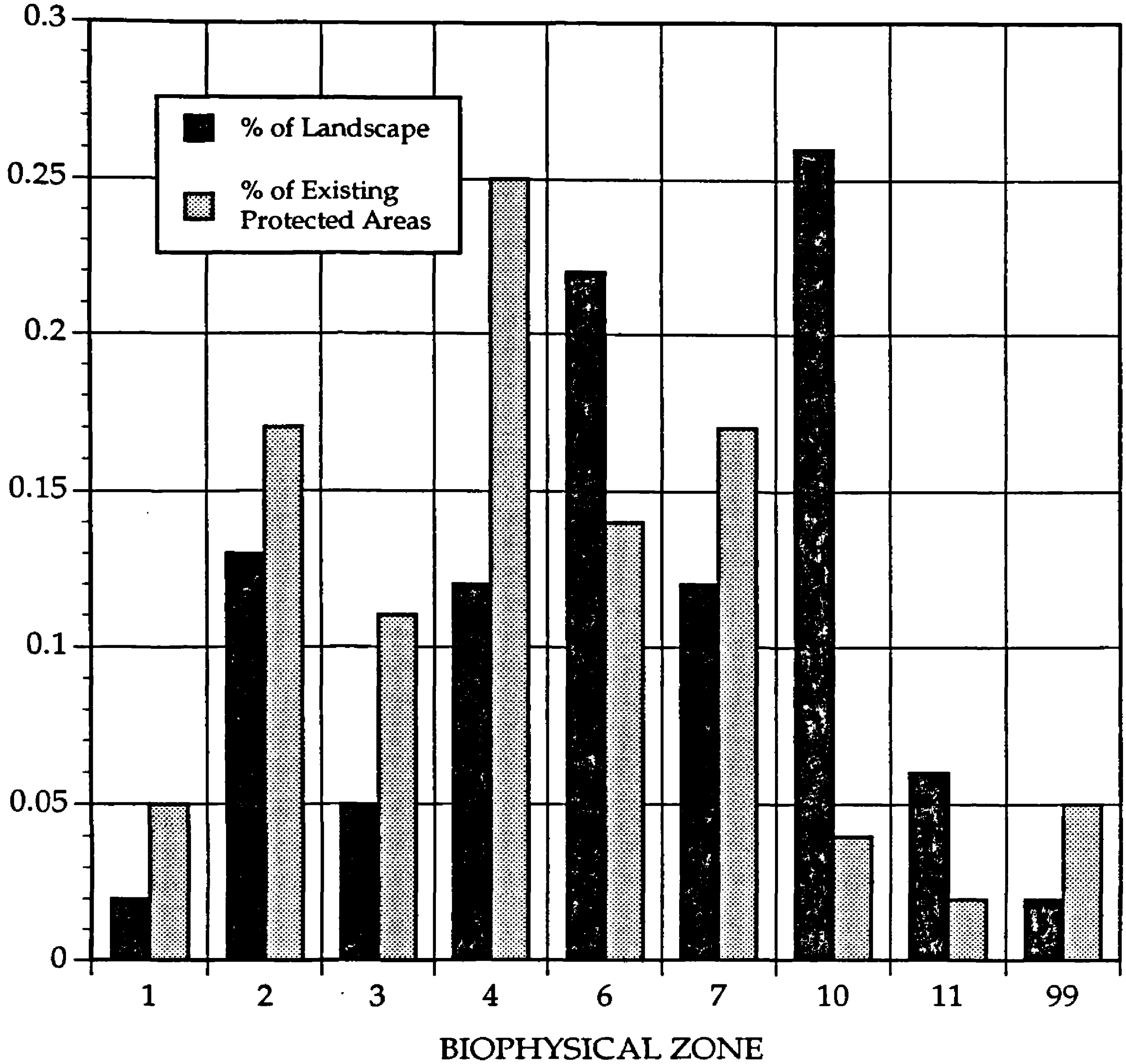


Figure 5-13. Representation of biophysical zones within existing protected areas (Levels 111 and 210) in relation to the proportion of the landscape occupied by each zone, Seeley-Swan landscape, northwestern Montana.

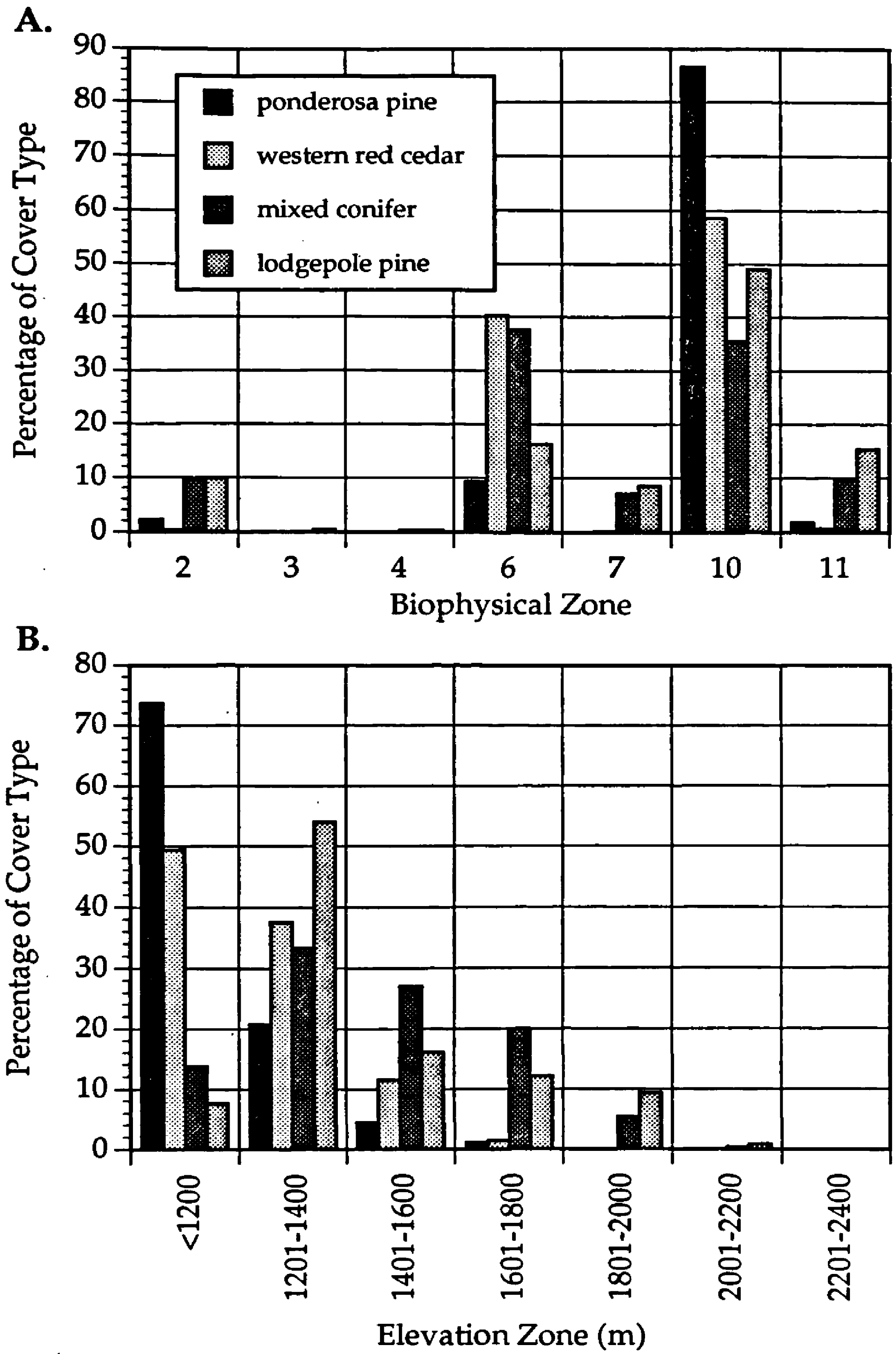


Figure 5-14. Distribution of underrepresented cover types by A) biophysical zone and B) elevation zone, Seeley-Swan landscape, northwestern Montana.

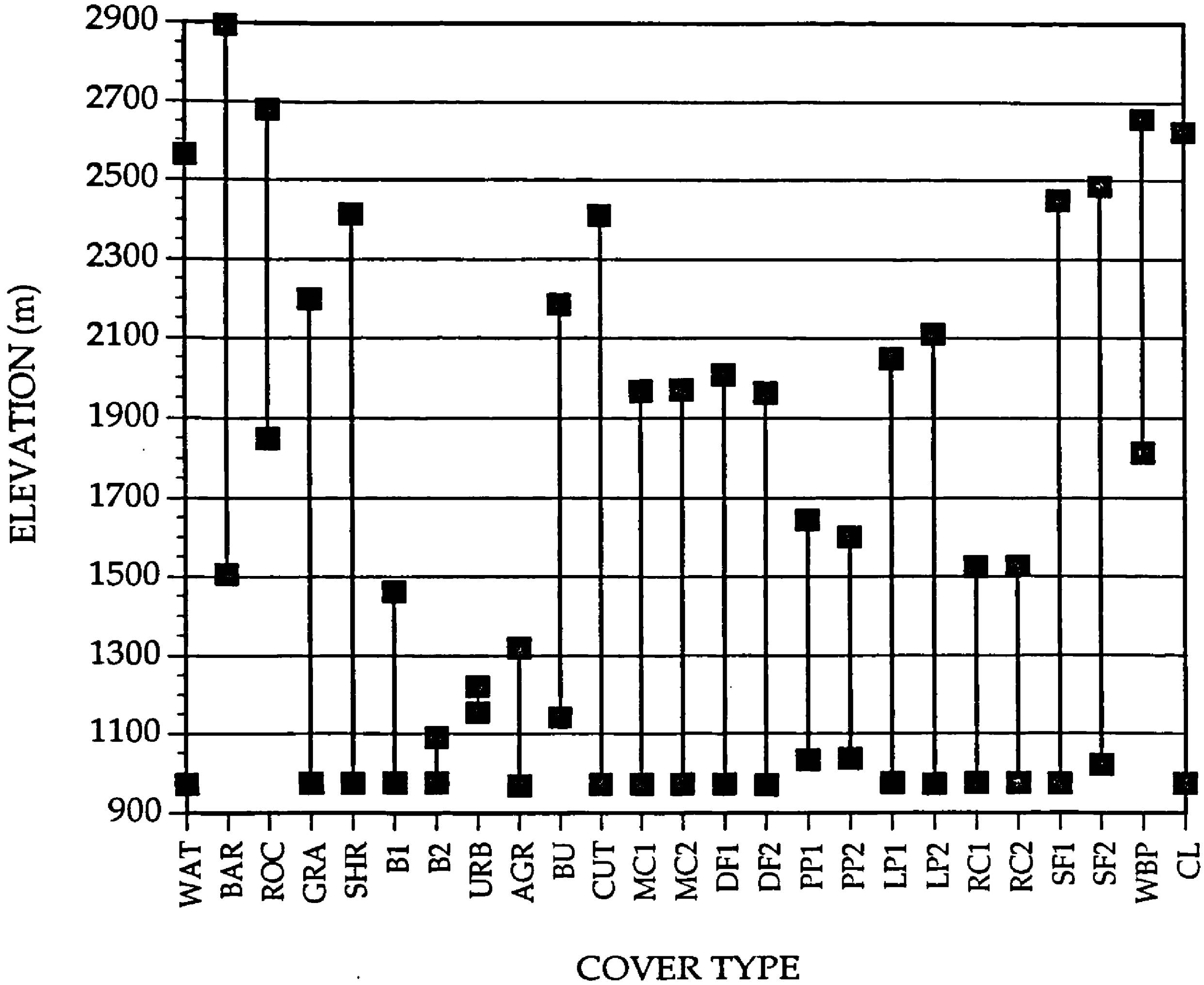


Figure 5-15. Elevation ranges for standardized cover types (see Table 3-3) in the Seeley-Swan landscape, northwestern Montana.

Table 5-8. 168 element occurrences, including species of special concern and other unique features, within the Seeley-Swan landscape, northwestern Montana (Natural Heritage Database, September 1994).

| Occurrences | Common Name | Scientific Name |
|---|---|---|
| 2 | Adder's-tongue | *Ophioglossum vulgatum* |
| 1 | Bald eagle | *Haliaeetus leucocephalus* |
| 1 | Beaked spikerush | *Eleocharis rostellata* |
| 1 | Beck water-marigold | *Bidens beckii* |
| 1 | Black swift | *Cypseloides niger* |
| 1 | Black tern | *Chlidonias niger* |
| 2 | Blunt-leaved pondweed | *Potamogeton obtusifolius* |
| 4 | Boreal owl | *Aegolius funereus* |
| 2 | Buckler fern | *Dryopteris cristata* |
| 1 | Bull trout | *Salvelinus confluentus* |
| 4 | Cliff toothwort | *Cardamine rupicola* |
| 4 | Clustered lady's-slipper | *Cypripedium fasciculatum* |
| 9 | Common loon | *Gavia immer* |
| 2 | Elrod's snail | *Oreohelix elrodi* |
| 6 | English sundew | *Drosera anglica* |
| 1 | Flat-leaved bladderwort | *Utricularia intermedia* |
| 1 | Fringed onion | *Allium fibrillum* |
| 6 | Giant helleborine | *Epipactis gigantea* |
| 5 | Green-keeled cottonsedge | *Eriophorum viridicarinatum* |
| 8 | Howell's gum-weed | *Grindelia howellii* |
| 3 | Kidney-leaf white violet | *Viola renifolia* |
| 1 | Kruckeberg's sword-fern | *Polystichum kruckebergii* |
| 5 | Loesel's twaydlade | *Liparis loeselii* |
| 7 | Mission mountain kittentails | *Synthyris canbyi* |
| 1 | Montana arctic grayling | *Thymallus arcticus montanus* |
| 1 | Mountain moonwort | *Botrychium montanum* |
| 1 | Northern bog clubmoss | *Lycopodium inundatum* |
| 5 | Pale sedge | *Carex livida* |
| 4 | Poor sedge | *Carex paupercula* |
| 1 | Slender wintergreen | *Gaultheria ovatifolia* |
| 7 | Small yellow lady's-slipper | *Cypripedium calceolus var parviflorum* |
| 1 | Small-headed tarweed | *Madia minima* |
| 3 | Sparrow's-egg lady's-slipper | *Cypripedium passerinum* |
| 1 | Spoon-leaf moonwort | *Botrychium spathulatum* |
| 2 | State champion tree | |
| 1 | Water bulrush | *Scirpus subterminalis* |
| 58 | Water howellia | *Howellia aquatilis* |
| 2 | Watershield | *Brasenia schreberi* |
| 1 | Western hemlock/queen's cup plant association | *Tsuga heterophylla/Clintonia uniflora* plant association |
| 1 | Western red cedar/devil's club plant association | *Thuja plicata/Oplopanax horridum* plant association |

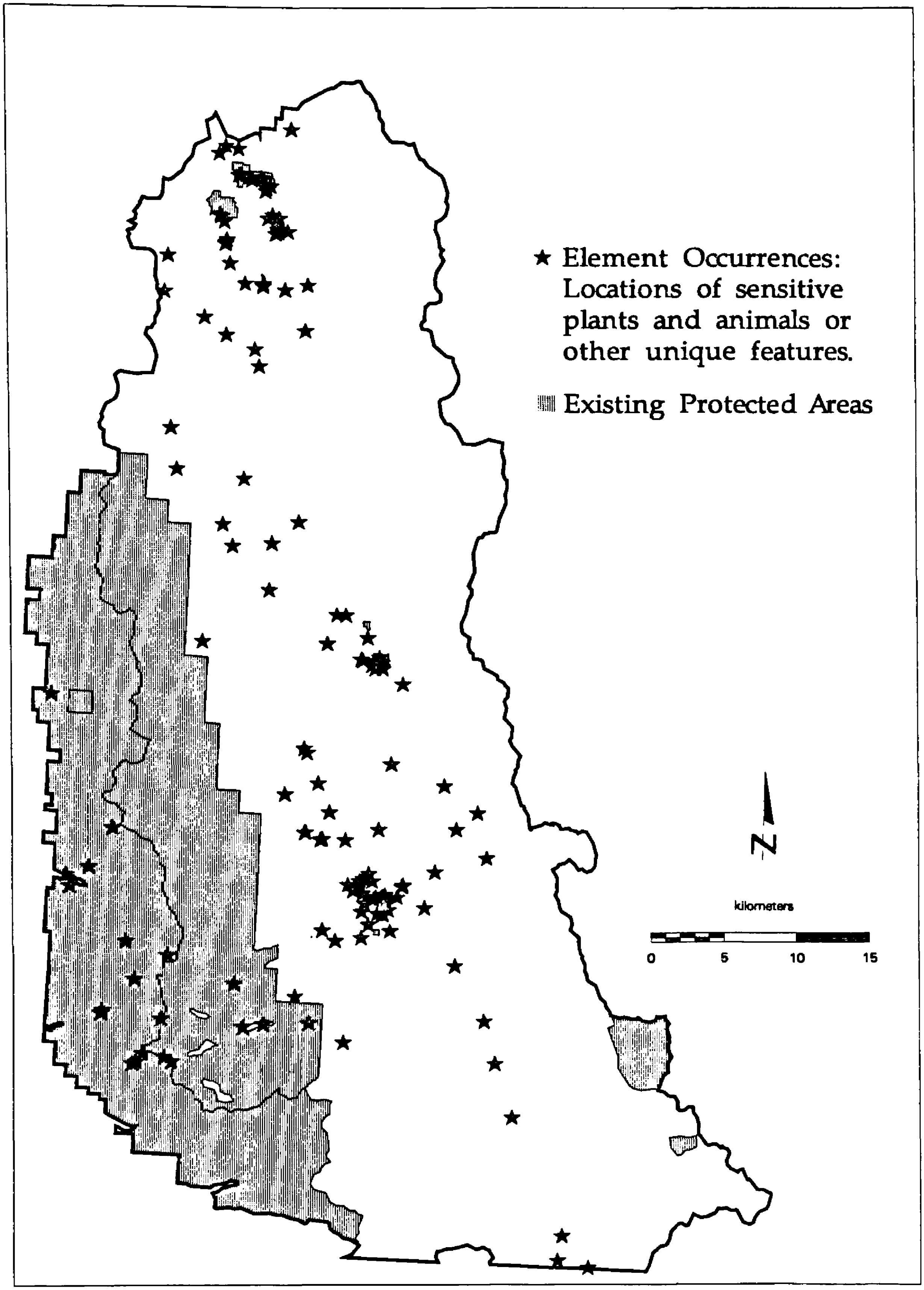


Figure 5-16. Location of 168 element occurrences from the Montana Natural Heritage Database in relation to existing protected areas, Seeley-Swan landscape, northwestern Montana.

Table 5-9. Element occurrences by protection level for the Seeley-Swan landscape, northwestern Montana. Element occurrences were acquired from the Montana Natural Heritage Database (September 1994), and include locations of sensitive plants, animals, and other unique features. Locations falling in water are for the common loon (*Gavia immer*).

| PROTECTION LEVEL | # ELEMENT OCCURRENCES | (%) |
|---|---|---|
| 111 | 48 | (29%) |
| 121 | 2 | (1%) |
| 122 | 19 | (11%) |
| 131 | 40 | (24%) |
| 132 | 2 | (1%) |
| 210 | 3 | (2%) |
| 220 | 1 | (<1%) |
| 230 | 47 | (28%) |
| water | 6 | (4%) |
| TOTAL | 168 | (100%) |

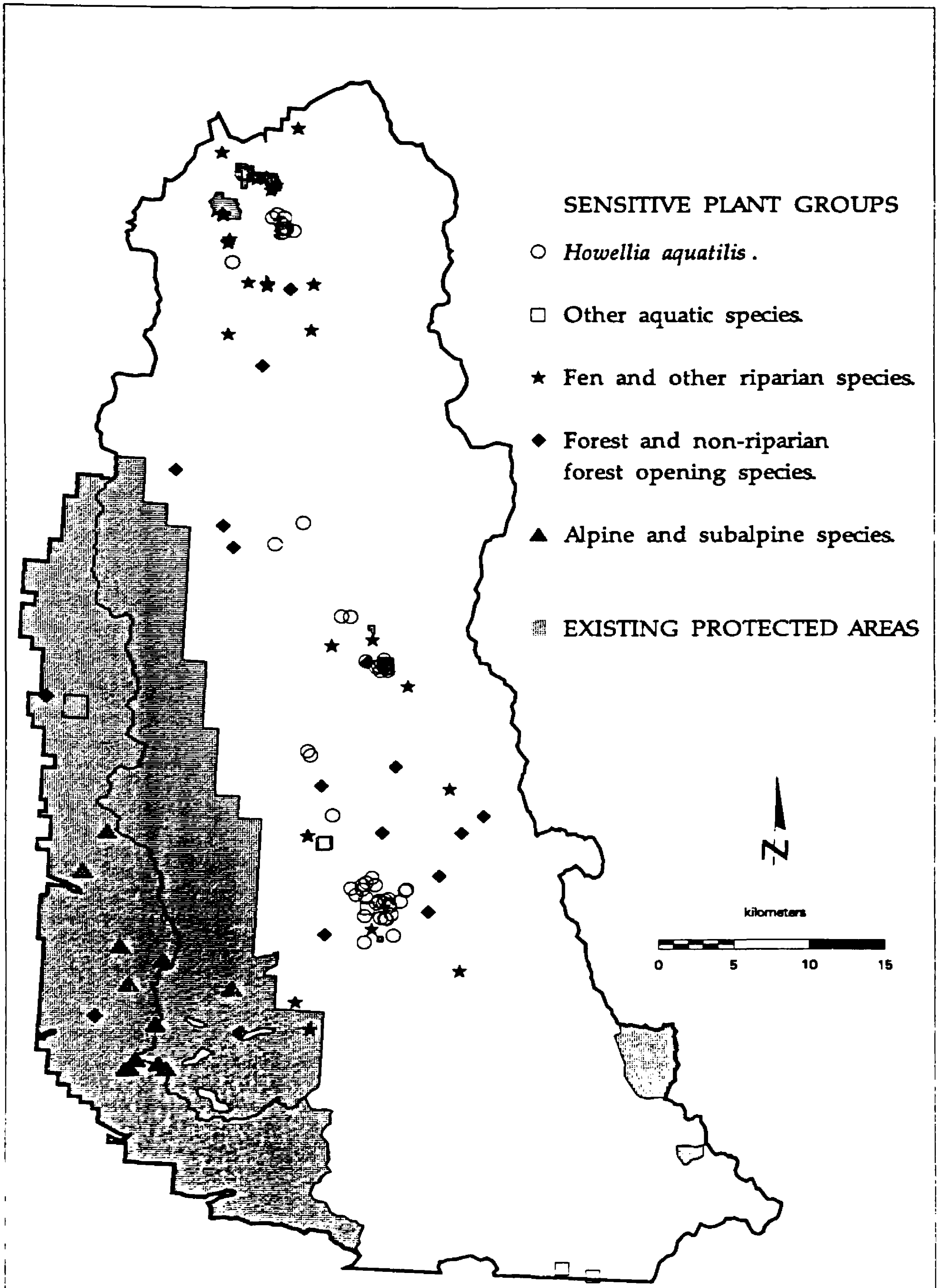


Figure 5-17. Locations of sensitive plants by habitat group, Seeley-Swan landscape, northwestern Montana. (Plant locations obtained from MT Natural Heritage Program.)

Table 5-10. Proportion of predicted habitat in protected status for twenty wildlife species in the Seeley-Swan landscape, northwestern Montana. Percentages are given for existing protected areas (Levels 111 and 210) as well as both alternatives proposed as supplements to the existing network.

| | TOTAL | % PROTECTED[a] | | |
|---|---|---|---|---|
| SPECIES | HABITAT (ha) | EXIST | ALT 1 | ALT 2 |
| Tailed frog | 2553 | 22.73 | 31.42 | 99.77 |
| Harlequin duck | 271 | 8.16 | 12.83 | 100.00 |
| Common loon | 1842 | 22.00 | 22.00 | 25.20 |
| Townsend's warbler | 55,636 | 26.61 | 37.66 | 66.69 |
| Black-backed woodpecker | 68,689 | 25.23 | 33.58 | 56.72 |
| Pileated woodpecker | 57,445 | 25.82 | 36.89 | 67.68 |
| Flammulated owl | 1767 | 1.57 | 13.45 | 99.58 |
| Boreal owl | 42,835 | 32.44 | 42.70 | 59.21 |
| Barred owl | 53,873 | 17.50 | 29.47 | 63.48 |
| Northern goshawk | 8312 | 23.78 | 30.66 | 50.71 |
| Bald eagle | 7379 | 1.09 | 9.27 | 77.05 |
| Peregrine falcon | 7571 | 31.40 | 31.92 | 42.92 |
| Marten | 45,536 | 25.16 | 36.46 | 57.53 |
| Fisher | 61,294 | 18.17 | 26.70 | 53.87 |
| Wolverine | 78,844 | 35.98 | 41.82 | 51.49 |
| Lynx | 28,741 | 24.26 | 30.82 | 37.80 |
| Gray wolf | 7013 | 0.04 | 7.74 | 32.77 |
| Grizzly bear | 122,781 | 55.46 | 60.41 | 63.44 |
| Mountain goat | 47,173 | 67.76 | 67.76 | 69.30 |
| Shiras moose | 55,603 | 18.05 | 25.90 | 56.00 |

[a] EXIST = existing, ALT 1 = alternative 1, ALT 2 = alternative 2.

Table 5-11. Full range of combinations included in scores rating sites for potential inclusion in a network of protected areas, Seeley-Swan landscape.

| Score | NP[a] | COV | ELEV | MOM | RD | P1 | P2 | P3 | P4 | P5 | Ha (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 0 | 1311 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12113 (4.8) |
| 1 | 9592 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 72800 (29.4) |
| 1 | 331 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 4709 (1.9) |
| 1 | 6066 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 60671 (24.5) |
| 1 | 131 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 880 (0.4) |
| 2 | 11 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 1094 (0.4) |
| 2 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 20 (<0.1) |
| 2 | 1462 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 21718 (8.8) |
| 2 | 9 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 360 (0.1) |
| 2 | 14 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1727 (0.7) |
| 2 | 2 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 10 (<0.1) |
| 2 | 17 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 2341 (0.9) |
| 2 | 1233 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 10884 (4.4) |
| 2 | 824 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 10566 (4.3) |
| 2 | 195 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1615 (0.7) |
| 2 | 208 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 2826 (1.1) |
| 2 | 669 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8371 (3.4) |
| 3 | 1 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 79 (<0.1) |
| 3 | 2 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 43 (<0.1) |
| 3 | 1 | 0 | 1 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 3 (<0.1) |
| 3 | 1 | 0 | 1 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 18 (<0.1) |
| 3 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 20 (<0.1) |
| 3 | 2 | 0 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 10 (<0.1) |
| 3 | 242 | 0 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 3372 (1.4) |
| 3 | 308 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 3782 (1.5) |
| 3 | 1 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 5 (<0.1) |
| 3 | 4 | 1 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1267 (0.5) |
| 3 | 106 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 641 (0.3) |
| 3 | 1566 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 15624 (6.3) |
| 4 | 1 | 0 | 1 | 1 | 1 | 0 | 0 | 1 | 0 | 0 | 47 (<0.1) |
| 4 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 587 (0.2) |
| 4 | 3 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 73 (<0.1) |
| 4 | 5 | 1 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 491 (0.2) |
| 4 | 4 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 423 (0.2) |
| 4 | 577 | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 8553 (3.4) |
| 5 | 1 | 1 | 1 | 1 | 1 | 0 | 0 | 1 | 0 | 0 | 190 (0.1) |
| Total | 24903 | 15 | 25 | 17 | 16 | 5 | 3 | 6 | 5 | 1 | 247932 (100) |

[a] NP = number of patches; COV = underrepresented cover type; ELEV = elevation <1600 m; MOM = mature/overmature forest; RD = road density ≤2.0 mi/mi$^2$; P1 = *Howellia aquatilis*; P2 = other aquatic plants; P3 = fen and other riparian plants; P4 = forest and non-forest opening plants; P5 = alpine and subalpine plants; Ha (%) = area and percentage of landscape for each combination.

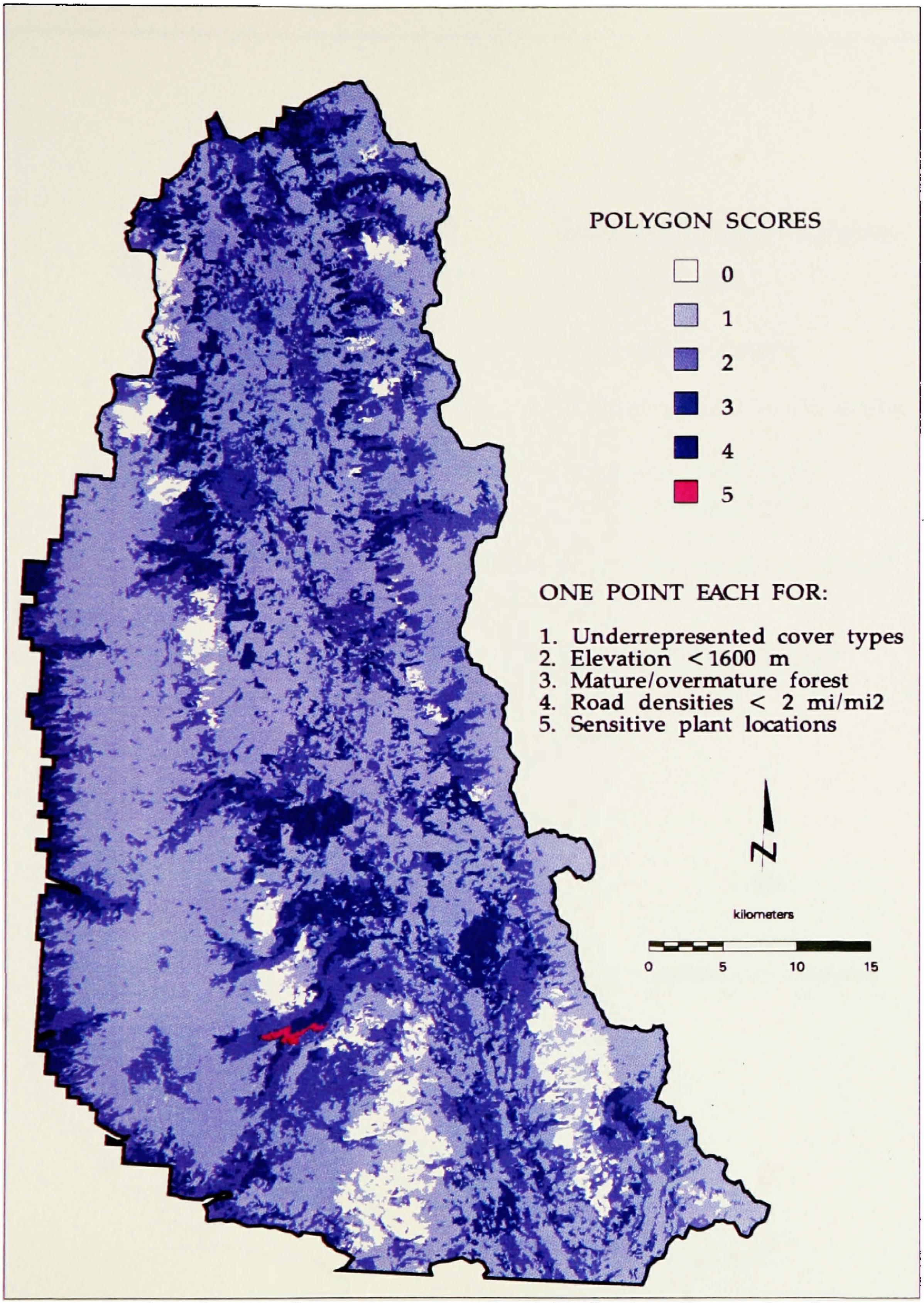


Figure 5-18. Scores for reserve potential, Seeley-Swan landscape, northwestern Montana. Only one polygon received the maximum observed score of 5.

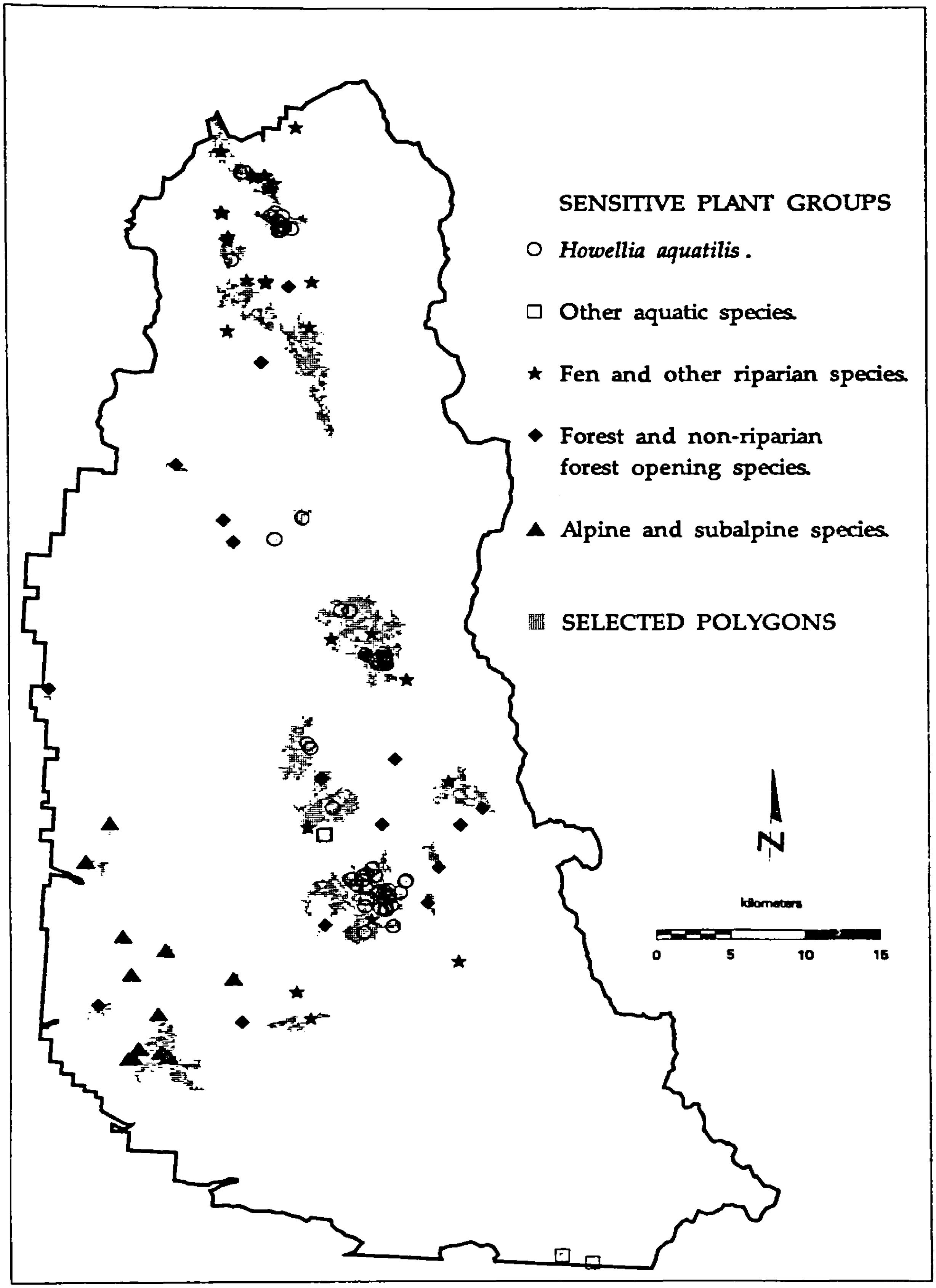


Figure 5-19. All polygons containing at least one sensitive plant species occurrence, Seeley-Swan landscape, northwestern Montana.

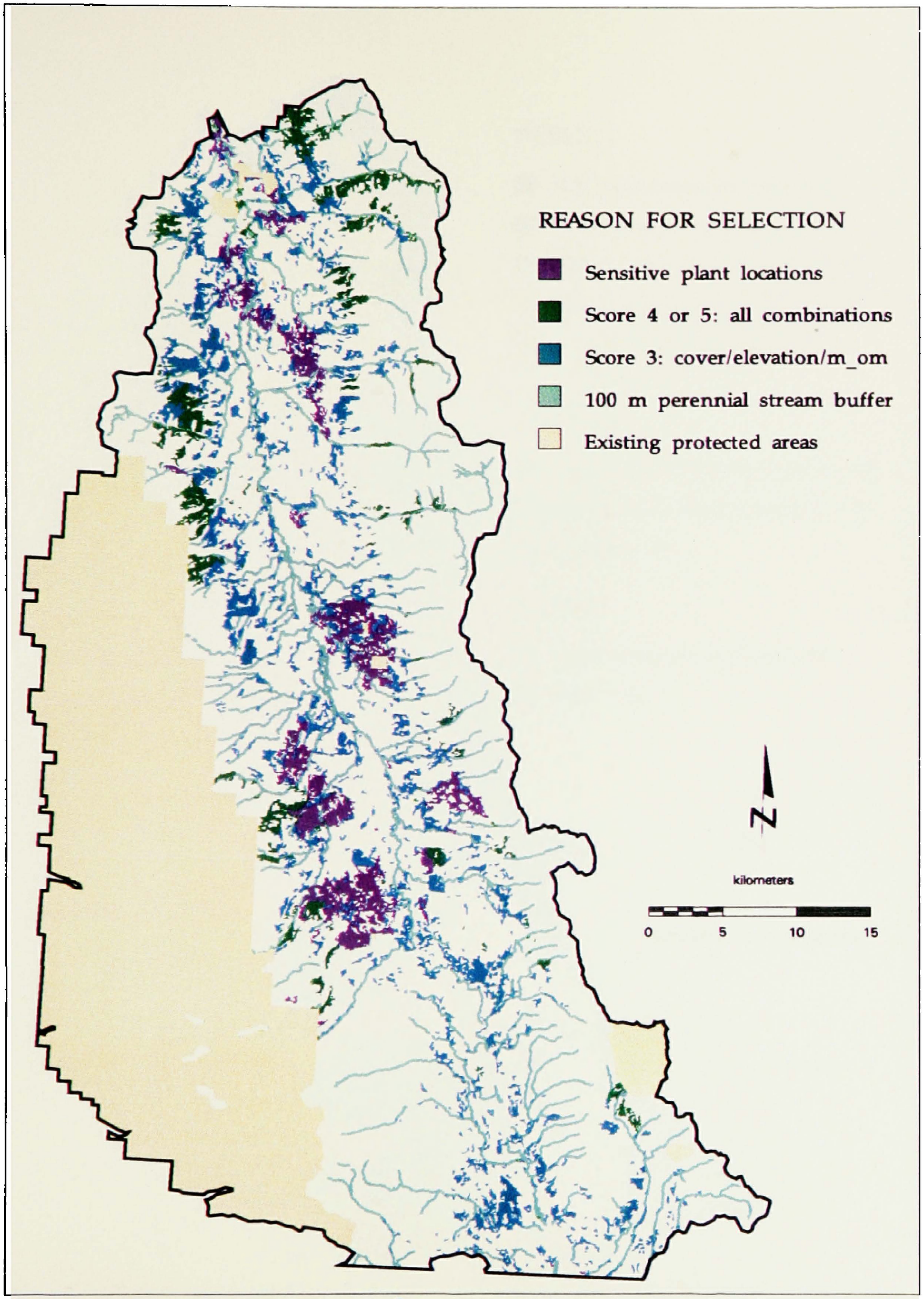


**Figure 5-20.** Sites targeted for potential inclusion in the existing reserve network and reasons for selection, Seeley-Swan landscape, northwestern Montana.

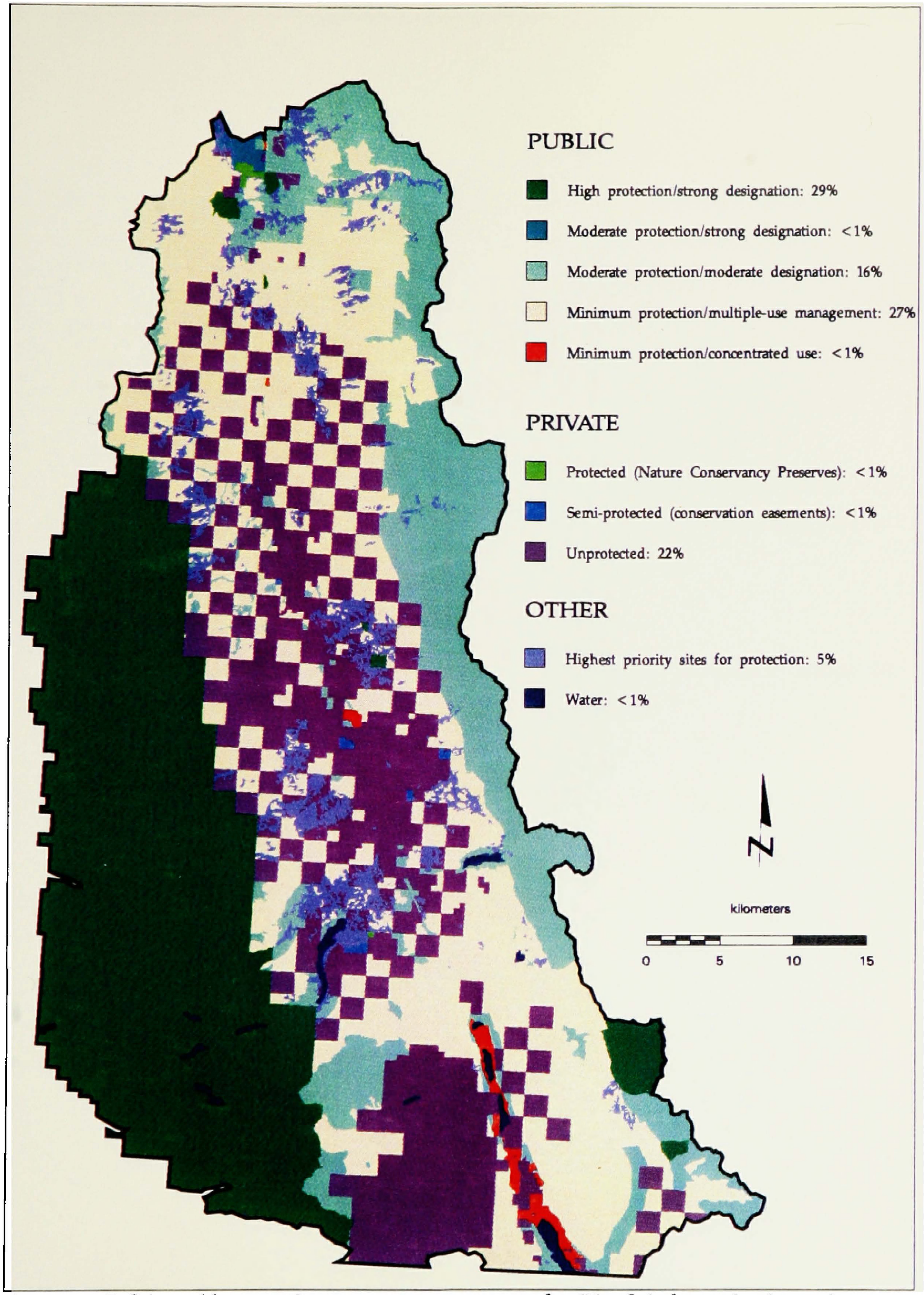


Figure 5-21. Alternative reserve network #1: high priority sites overlayed with existing management and protection designations in the Seeley-Swan landscape, northwestern Montana.

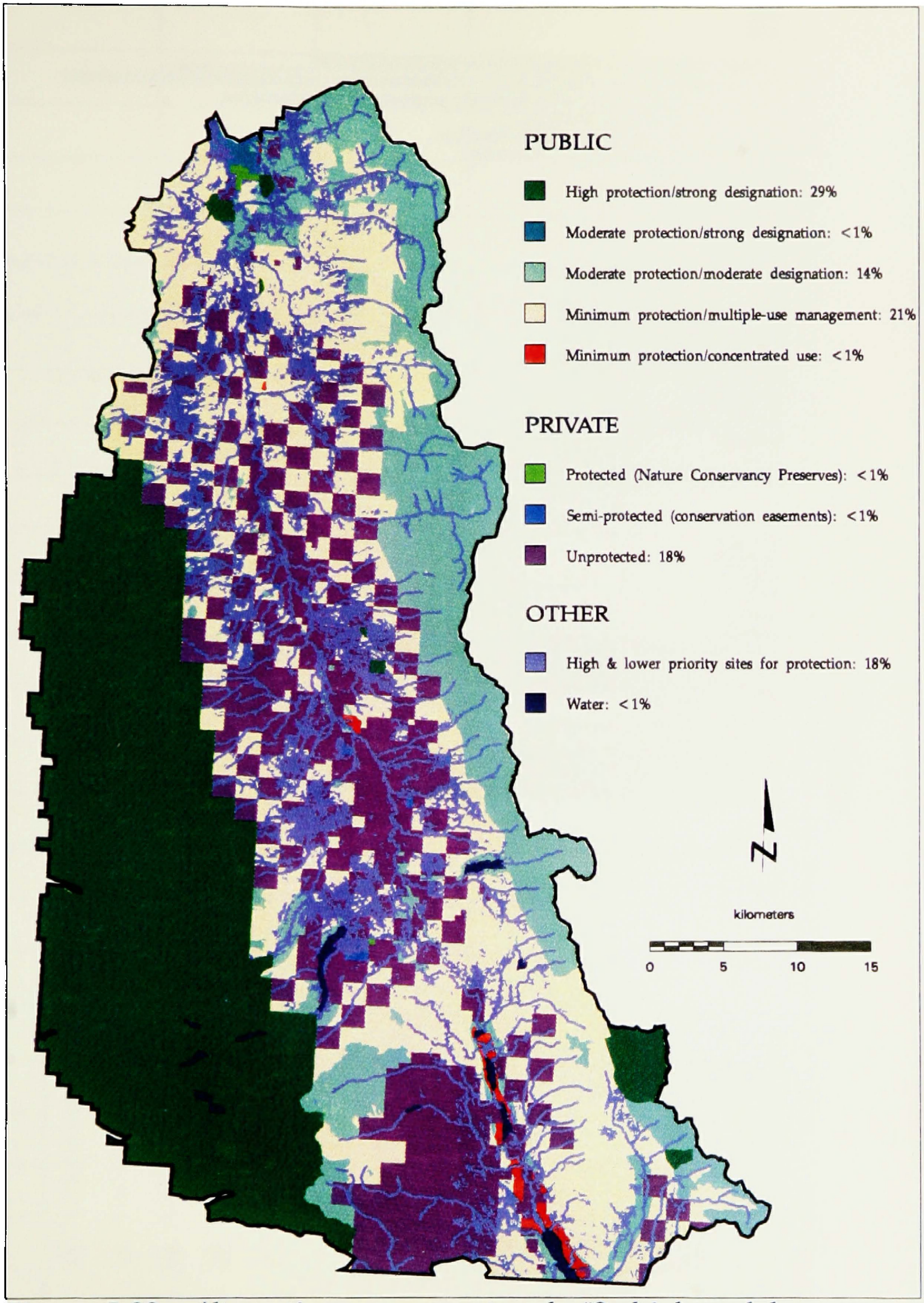


Figure 5-22. Alternative reserve network #2: high and lower priority sites overlayed with existing management and protection designations in the Seeley-Swan landscape, northwestern Montana.

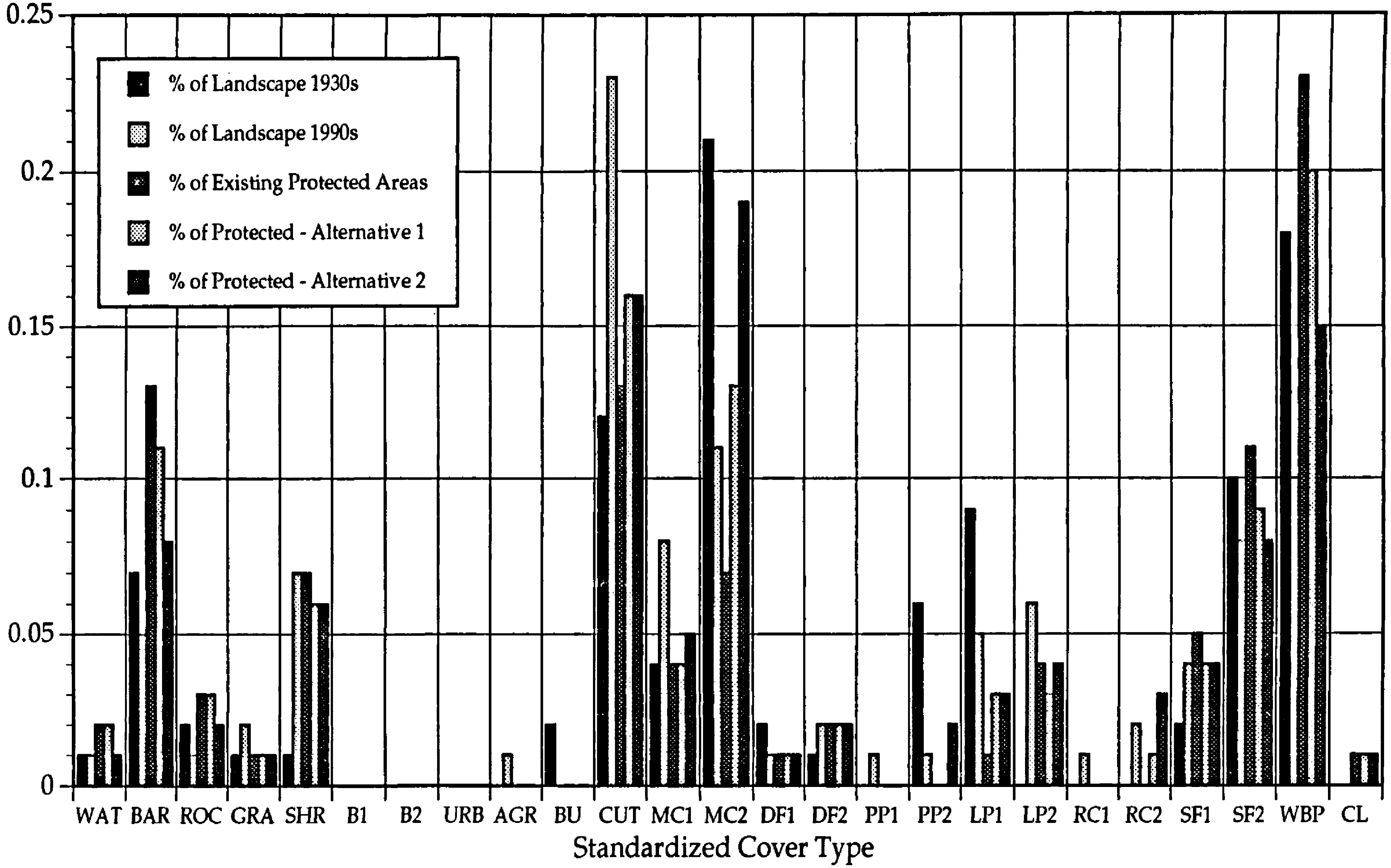


Figure 5-23. Proportion of each cover type in total landscape and in existing and proposed reserve networks, Seeley-Swan landscape, northwestern Montana.

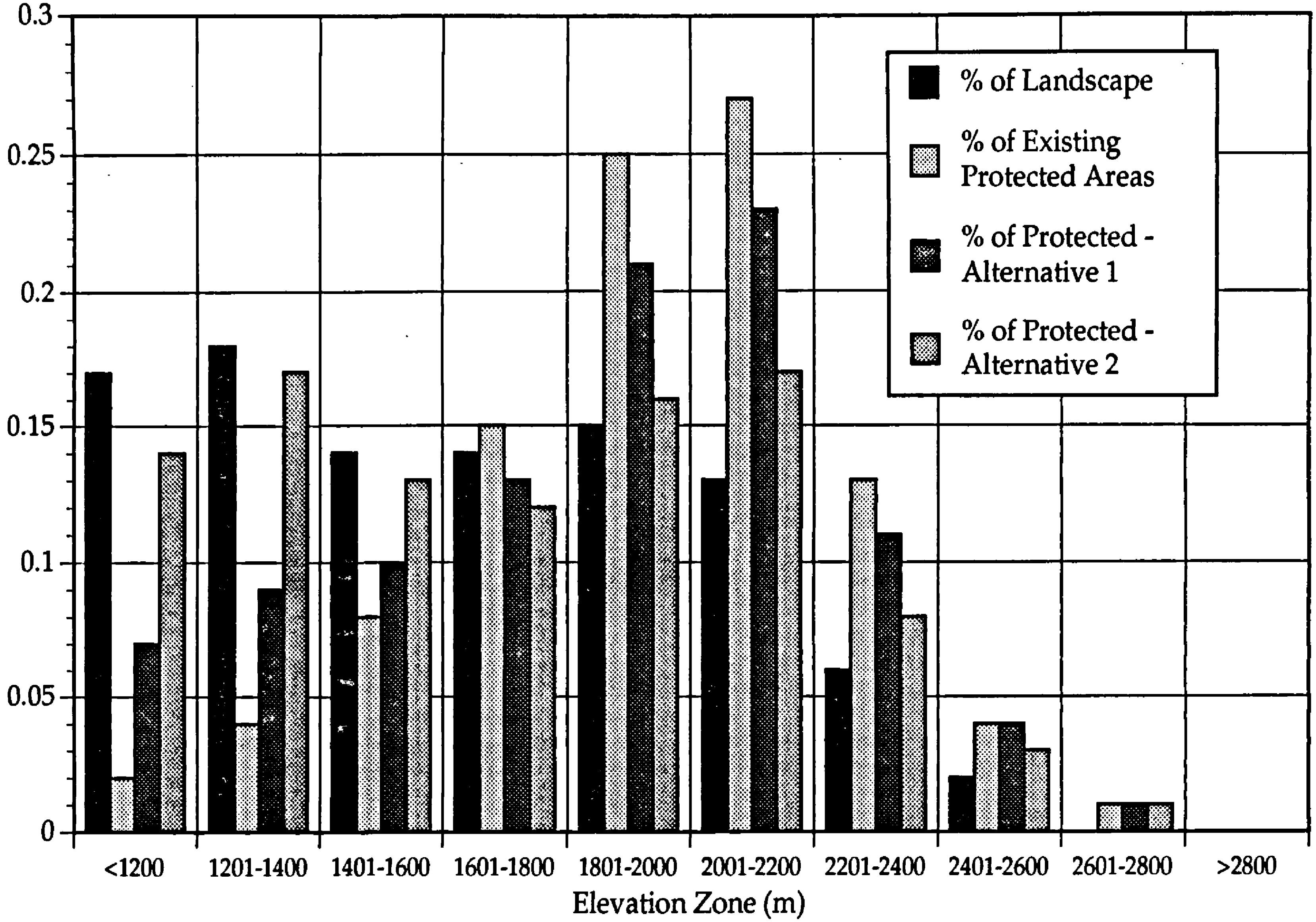


Figure 5-24. Proportion of each elevation zone in the total landscape and in existing and proposed reserve networks, Seeley-Swan landscape, northwestern Montana.

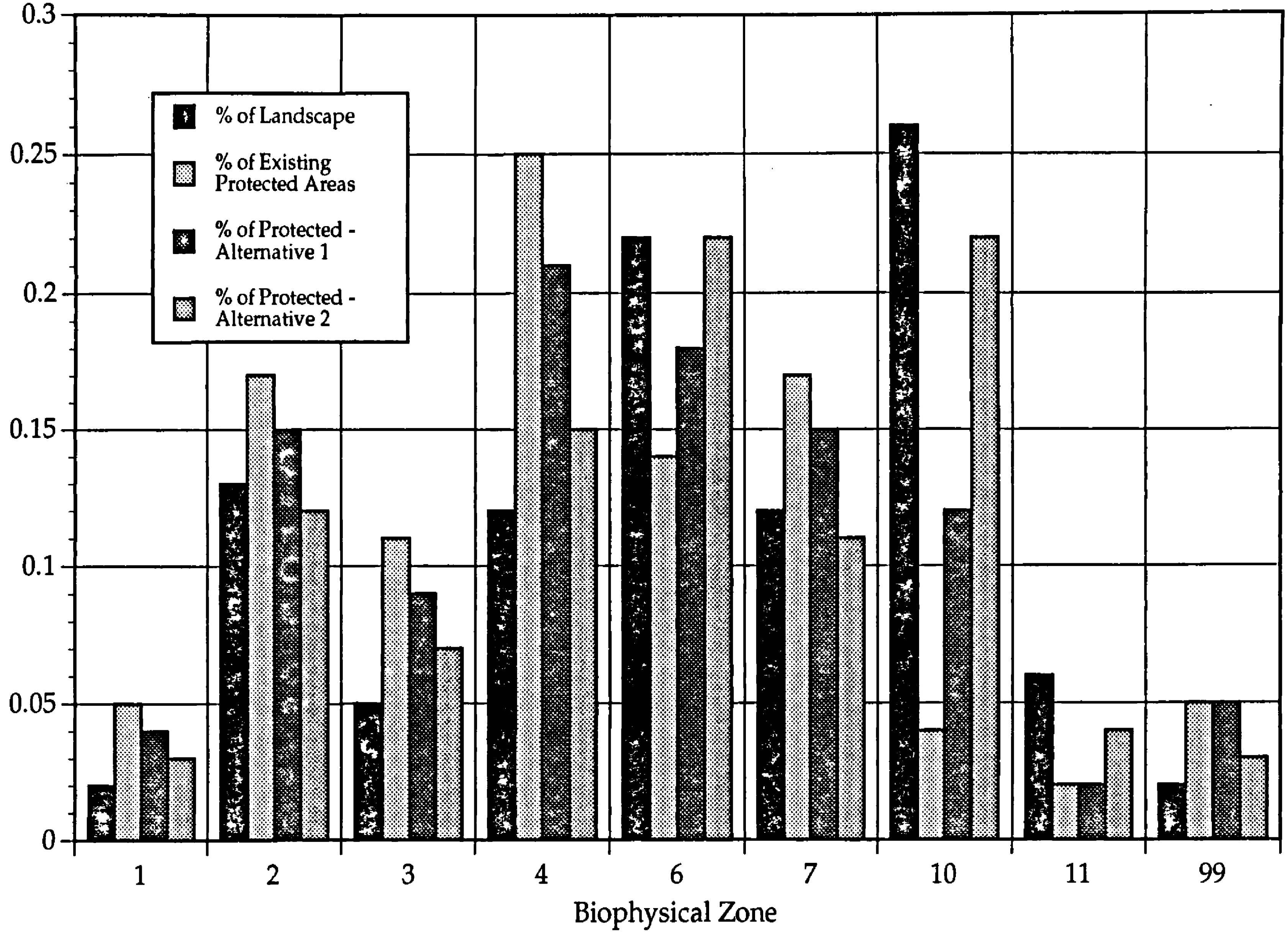


Figure 5-25. Proportion of each biophysical zone in the total landscape and in existing and proposed reserve networks, Seeley-Swan landscape, northwestern Montana.

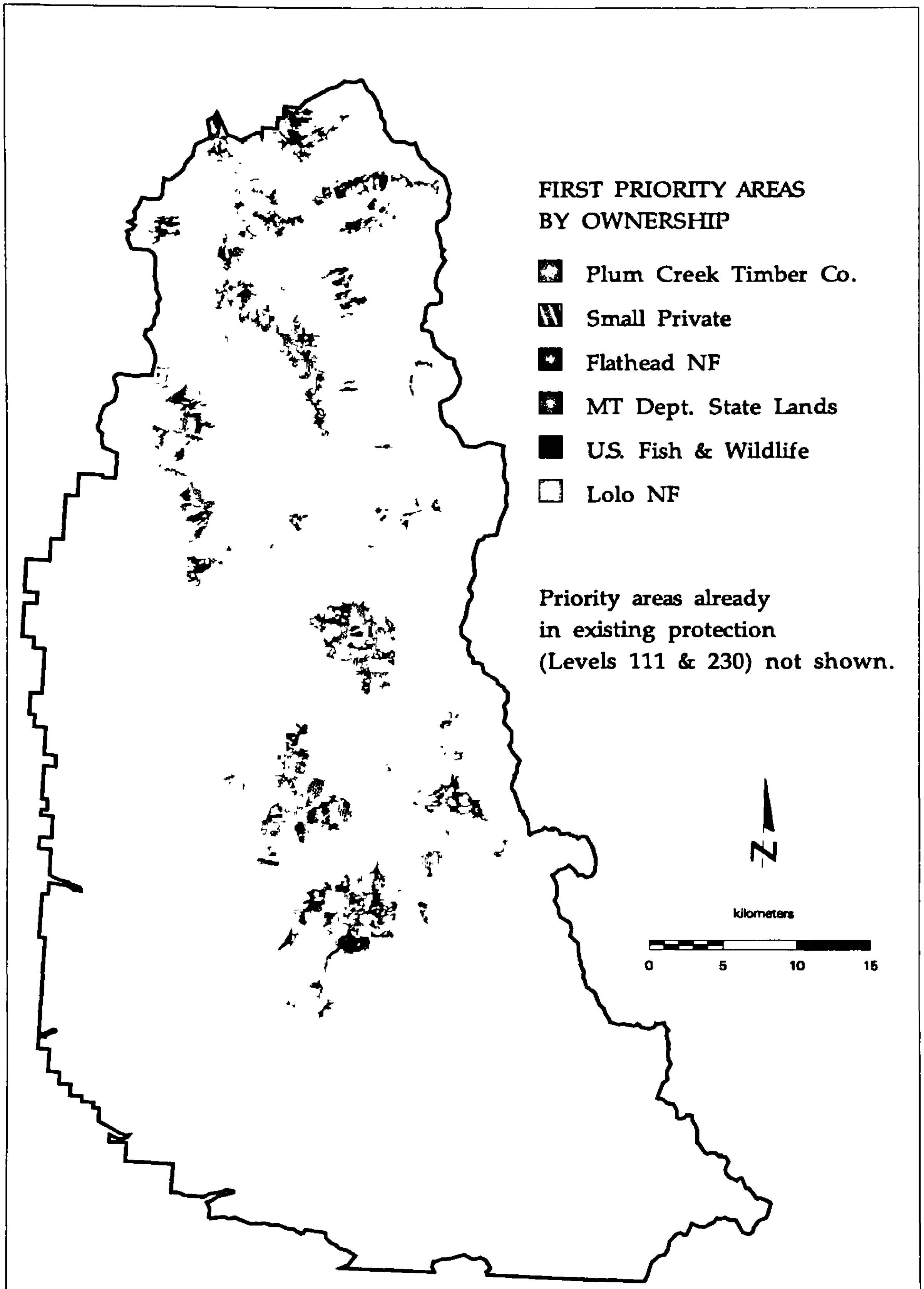


Figure 5-26. First priority sites targeted in the reserve selection process in relation to land ownership, Seeley-Swan landscape, northwestern Montana.

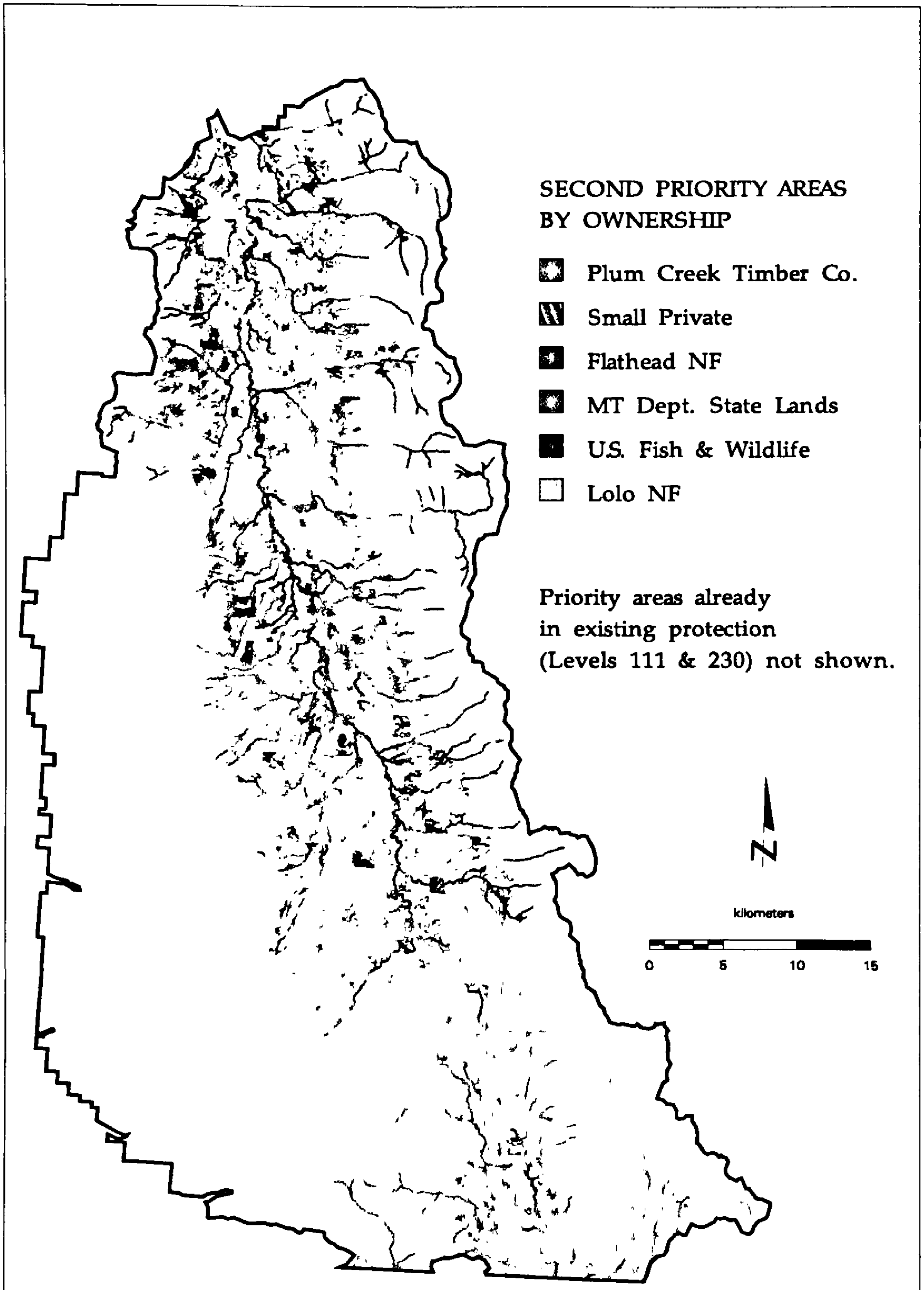


Figure 5-27. Second priority sites targeted in the reserve selection process in relation to land ownership, Seeley-Swan landscape, northwestern Montana.

Table 5-12. Ownership and management of first and second priority sites for supplementation of the existing network of protected areas, Seeley-Swan landscape, northwestern Montana. First priority areas occupy 12,488 ha, and second priority areas cover 32,116 ha.

| OWNER | % PRIORITY 1 | % PRIORITY 2 |
|---|---|---|
| **PUBLIC:** | | |
| Flathead NF[a] | 43 | 33 |
| Lolo NF[b] | 2 | 17 |
| U.S. Fish & Wildlife Service | 1 | <1 |
| Montana Dept. of State Lands | 12 | 12 |
| **PRIVATE:** | | |
| Plum Creek Timber Co. | 32 | 27 |
| Small Private | 9 | 11 |

[a] Priority 1: 36% currently under Level 122 management; 64% Level 131. Priority 2: 37% Level 122; 62% Level 131.

[b] Priority 1: 3% Level 122; 93% Level 131; 4% Level 132. Priority 2: 17% Level 122; 75% Level 131; 8% Level 132.

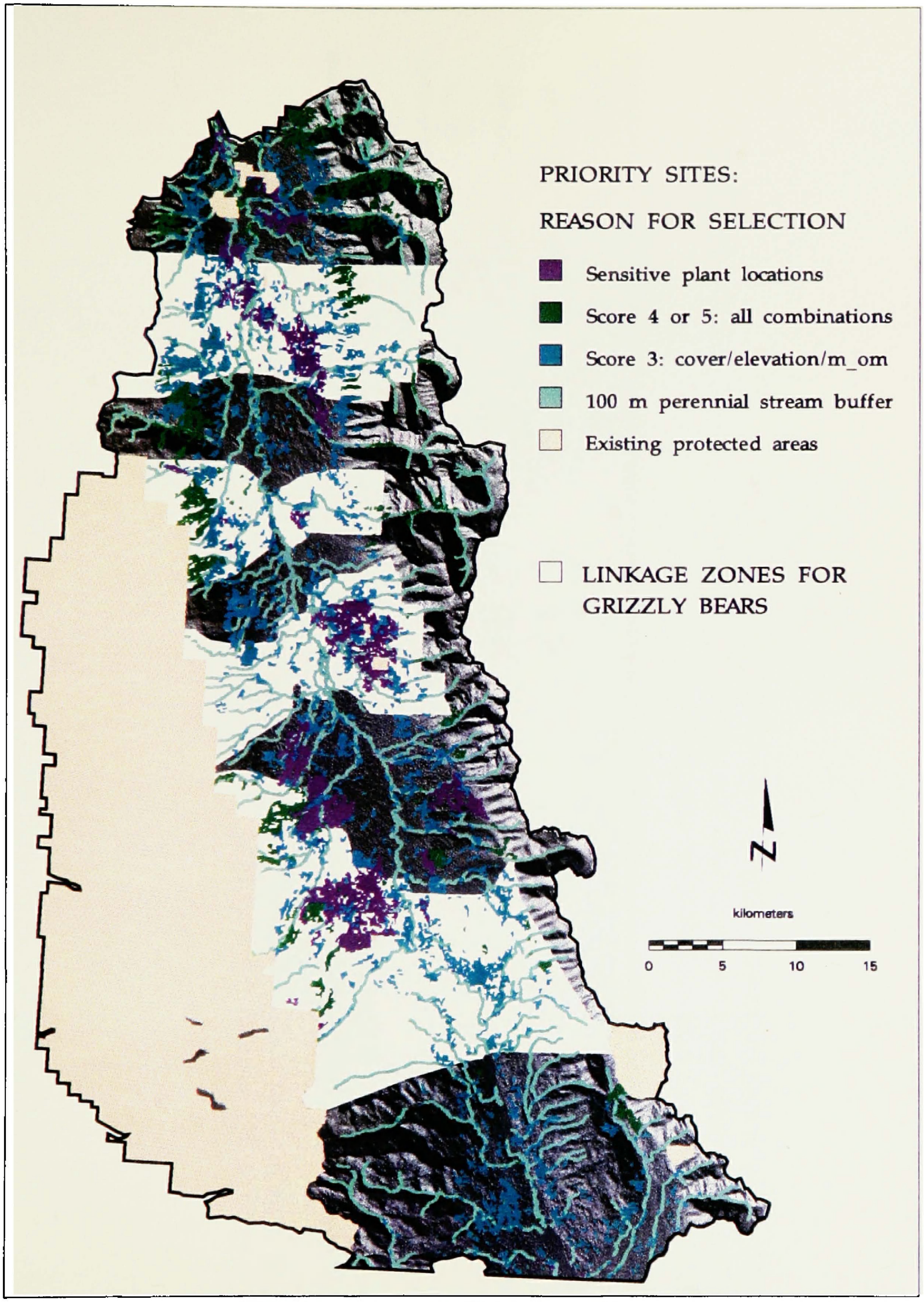


Figure 5-28. Linkage zones for grizzly bears (Servheen and Sandstrom 1993) in relation to sites identified as potential additions to the existing reserve network, Seeley-Swan landscape, northwestern Montana.

Table 5-13. Linkage zones for grizzly bears (Servheen and Sandstrom 1993) in relation to sites selected for potential inclusion in the existing network of protected areas, Seeley-Swan landscape, northwestern Montana.

| LINKAGE ZONE[a] | AREA (ha) | % PLANT[b] | % SCORE ≥4 | % SCORE 3 | % BUFFER | SUM |
|---|---|---|---|---|---|---|
| 1 | 17,393 | 6.81 | 2.48 | 13.15 | 8.31 | 30.75 |
| 2 | 8732 | 1.57 | 8.89 | 12.33 | 6.89 | 29.68 |
| 3 | 11,635 | 9.74 | 0.16 | 8.71 | 14.86 | 33.47 |
| 4 | 30,027 | 5.88 | 3.35 | 7.90 | 7.77 | 24.90 |

[a] Numbered sequentially from north to south; see Figure 5-28.
[b] Percentage of linkage zone occupied by areas selected for sensitive plants, score ≥4, score 3, and riparian buffers. SUM = total percentage of linkage zone occupied by selected areas.

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# APPENDIX A:
## Assumptions in Standardizing Cover Type Codes

The following assumptions were made in regrouping 1930s and 1990s cover types to standardized codes (Table 3-3):

BARREN: The 1930s barren type matched well in an on-screen comparison with a combination of the 1990s snow, melted snow, rock, and barren types; thus, the four 1990s types were lumped into the barren type.

ROCKY WOODLAND: For the 1930s, woodland types were grouped with rocky noncommercial types because both are noncommercial, and both contain similar tree species at low density.

GRASS: 1990s grass and wet meadow types were combined under the assumption that the 1930s grass type included wet meadows: the 1930s legend did not include a wet meadow type, and the spatial arrangement of grass in the 1930s coincided with my knowledge of the current distribution of wet meadows in the Seeley-Swan.

SHRUB: The 1990s grass/ shrub type was placed in the shrub category simply because I opted for the taller life form.

RECENT CUT/ SEEDLING/ SAPLING: Recent cuts were grouped with seedlings and saplings because I could not reliably distinguish recent cuts from seedling plantations using Landsat TM data. Seedlings and saplings were combined because they were grouped as one stand class in the 1930s data set.

PONDEROSA PINE: The 1930s Douglas-fir/ ponderosa pine type was included with ponderosa pine based on species composition data, which revealed a percentage of ponderosa pine in the stands comparable with the 1990s definition.

MIXED CONIFER: Perhaps the greatest liberty was taken with the mixed conifer class. Along with the 1990s mixed conifer and mixed conifer/ broadleaf types, grand fir was included in this type because it tends to occur in mixed stands and because there is no corresponding 1930s grand fir type. Similarly, there is no 1930s mixed conifer type: I opted to include western larch/ Douglas-fir and western white pine in the mixed conifer type based primarily on species composition data for these stands. The 1930s western larch/ Douglas-fir type is extensive, as is the 1990s mixed conifer type, and they exhibit some spatial overlap. In the 1930s, western white pine was a preferred commercial species, and any stand with 15% or more of its volume in western white pine was typed as such (USDA:FS 1937), making it likely that many of the 1930s western white pine stands actually had mixed species composition. Unfortunately, species composition data was available for only one western white pine stand, but these data upheld that assumption.

CLOUD/ CLOUD SHADOW: The 1990s cloud and cloud shadow types cannot be compared with any 1930s types, but they are an unavoidable consequence of using satellite data for vegetation mapping. Fortunately, their areal extent is fairly small.

## APPENDIX B:
## Calculation of Road Densities using ARC/INFO

An ARC/INFO vector layer of all roads and trails for Swan Lake Ranger District, current as of 1994, was acquired from Flathead National Forest (USDA:FS 1994a). The vector layer was then converted from Universal Transverse Mercator (UTM) zone 12 to Albers conical equal-area projection and clipped to match the study area boundary. A new layer from which trails were eliminated was then created. Next, cartographic feature files for 7.5' quadrangles covering Seeley Lake Ranger District were acquired from the Forest Service (Northern Regional Office, Management Systems Unit), in MOSS format. These were converted to ARC/INFO vector files and appended to form a single layer. Again, trails were eliminated. Layers for the 2 districts were then appended to form one vector layer of all roads within the study area.

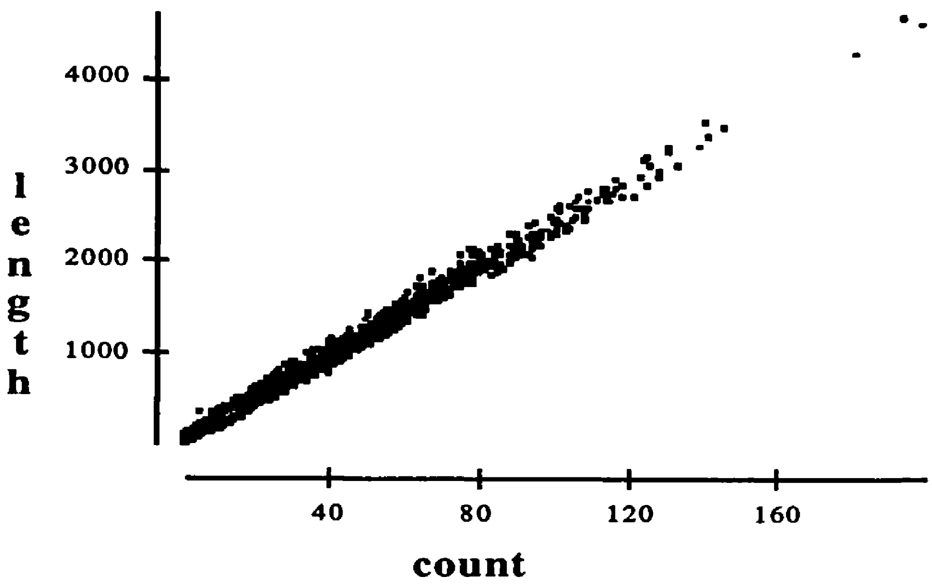
This layer was then converted to raster format at a 30 m cell size using the LINEGRID function in ARC/INFO; values in the raster layer were based on road-ID's from the vector database. The areal extent of the raster layer was then expanded by filling in null values so that road density calculations would not be sheared off at the edges of the layer. Next, each road cell was assigned a value of 1 in preparation for road density calculation.

Then, a moving-circle technique called FOCALSUM (ESRI 1991) was used to count the number of road cells within a 900 m (30-cell) search radius, and assign the sum to the corresponding cell in a new layer. The 900 m search radius was chosen to yield a circle with area as close as possible to 1 mi/mi$^2$. Given that 1 mi is about 2,590,000 m$^2$, and that a circle's area is PI * r$^2$, the radius of a circle with area = 1 mi$^2$ would be about 908 m. Thus, the most appropriate radius based on a cell size of 30 m would be 30 cells. Within this 30-cell-radius circle, there were 2828 cells, which converted to 0.98276 mi$^2$.

The next step was to classify cell counts into road densities (mi/mi$^2$) using the following formula: (1609.3 * 0.98276 * n) / 30, where n = number of mi/mi$^2$ and given 1609.3 m in 1 mi, an actual analysis area of 0.98276, and cell size (length) of 30 m. From this equation, the cutoff point for the number of cells equivalent to a given number of mi/mi$^2$ of road was determined; for example, 53 cells is equal to 1 mi/mi$^2$ of road. Once cutoff values were determined, cell counts were regrouped: 0-53 cells were assigned a value of 1, 54-105 cells a value of 2, and so on. In this manner, every cell in the newly-created layer was assigned a mi/mi$^2$ value. This conversion is based on the assumption that one 30 m cell in the raster layer is equivalent to 30 m of road length in the vector layer. This assumption is violated because of the conservative nature of the algorithm employed in the LINEGRID function (which is entirely appropriate for many situations), where an entire cell is labeled road if even the smallest portion of the vector road crosses it. Thus, road densities are overestimated.

Recognizing this, I applied a correction factor to my results. First, I related the databases for the vector and raster layers based on road-ID's, selected all records for which there was an entry in both databases, and exported the vector length (m) and the raster count (number of 30 m cells) along with the road-ID. I then regressed vector length on raster count (Figs. B-1, B-2), and generated a new variable with values matching the range of cell counts from the FOCALSUM function (about 1-1000). Using the regression equation, $y = 10.9750 + 23.7251x$, I predicted vector lengths for each cell count. I then adjusted the number of meters per $mi^2$ based on the actual analysis area (0.98276 $mi^2$), multiplied the adjusted value by the desired $mi/mi^2$ value, and looked for the closest match among the predicted lengths. The corresponding cell count was then used as a cutoff value in regrouping the cell counts as described earlier (Figs. B-3, B-4).

It should be noted that the LINEGRID function in ARC/INFO does not always lead to an overestimation of road length. A certain number of road segments disappear in the conversion from vector to raster, and are thus not included in the above determination of a correction factor. At road intersections, some differences in length are inevitable: Because each cell can only be assigned one road-ID, one segment will maintain or gain length and the other will lose it. For similar reasons, road lengths will also be underestimated in areas where roads are very closely spaced, including switchbacks. Thus, application of a correction factor may actually overcompensate for the LINEGRID algorithm, and lead to underestimation of road densities.




Dependent variable is: length
No Selector
R squared = 99.2% R squared (adjusted) = 99.2%
s = 46.73 with 6128 - 2 = 6126 degrees of freedom

| Source | Sum of Squares | df | Mean Square | F-ratio |
|---|---|---|---|---|
| Regression | 1585176123 | 1 | 1585176123 | 725986 |
| Residual | 13376004 | 6126 | 2183.48 | |

| Variable | Coefficient | s.e. of Coeff | t-ratio | prob |
|---|---|---|---|---|
| Constant | 10.9750 | 0.8671 | 12.7 | $\le 0.0001$ |
| count | 23.7251 | 0.0278 | 852 | $\le 0.0001$ |

Figure B-1. Regression of vector length (m) on raster cell count for the ARC/INFO road layer, Seeley-Swan landscape.

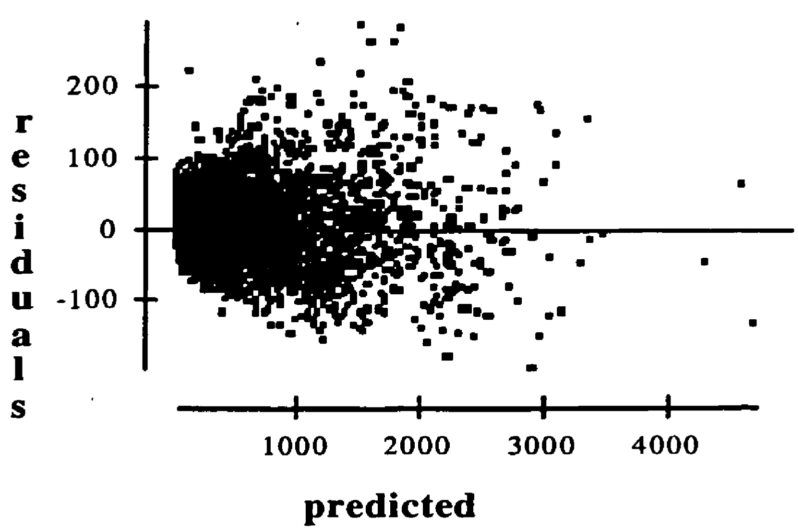



Figure B-2. Scatterplot of residuals versus predicted values for the above regression of vector length on cell count for roads.

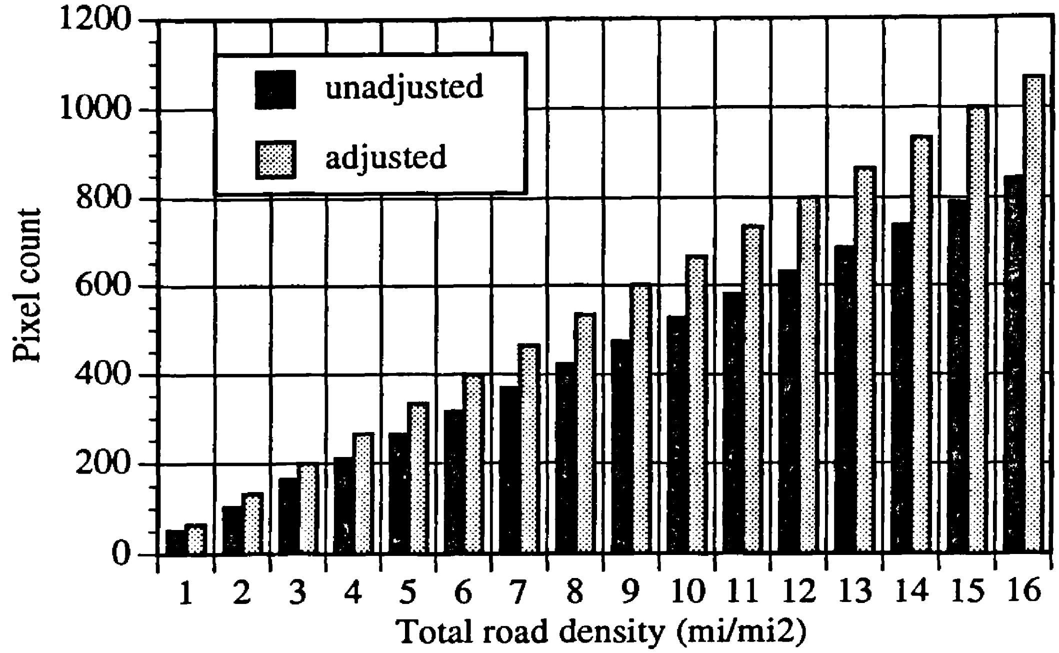



Figure B-3. Differences in cutoff values used to classify raw pixel counts into road density values (mi/mi2) before and after adjustment. Because vector road lengths are overestimated in the ARC/INFO conversion to raster format, adjustments were made by regressing vector road lengths on raster cell counts.

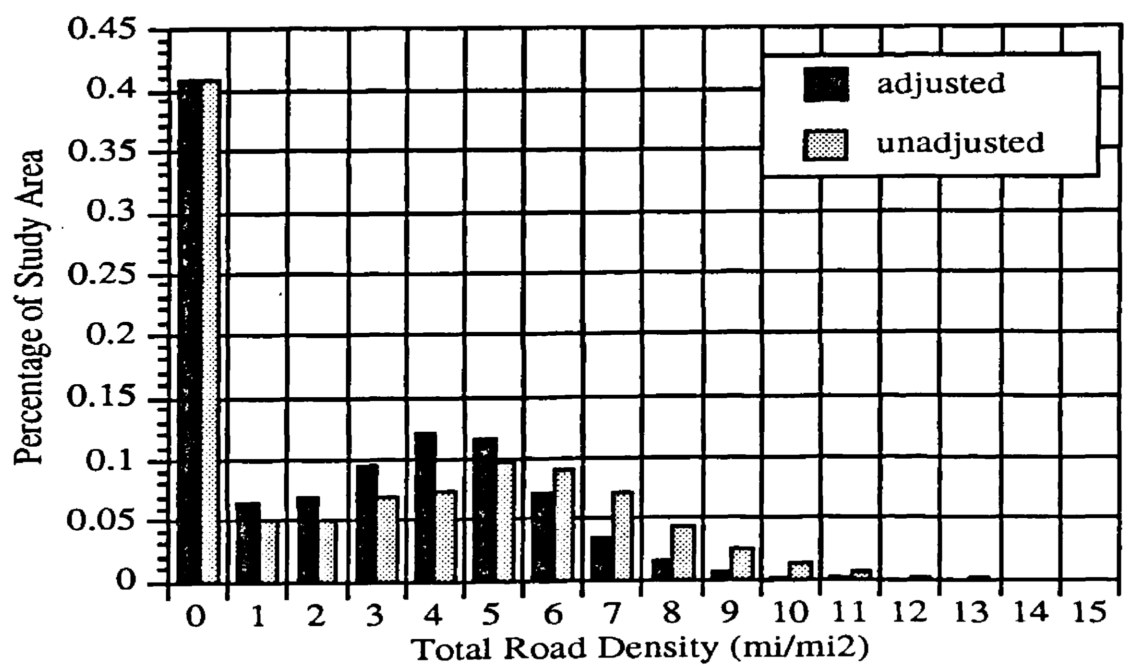



Figure B-4. Frequency distributions for total road density (mi/mi2) in the Seeley-Swan landscape before and after correction for overestimation of road lengths in the vector-to-raster ARC/INFO conversion process.

## APPENDIX C:
## Wildlife Habitat Descriptions, Modeling Rules, and Assumptions

### TAILED FROG
*(Ascaphus truei)*

HABITAT:

In the northern Rocky Mountains, the tailed frog occupies highly insular habitat, and very probably exists in fragmented, semi-isolated populations (Daugherty and Sheldon 1982). As a result, if populations are extirpated from a drainage, recolonization may be slow (Bury and Corn 1988). Tailed frogs inhabit small, cold, permanent, fast-flowing streams within forested areas (Metter 1964, Daugherty 1979, Daugherty and Sheldon 1982, Nussbaum et al. 1983, Hawkins et al. 1988). Stream gradients are typically high where tailed frogs are found (7%, Daugherty and Sheldon 1982; 15% and 35%, Metter 1964). In western Montana, tailed frogs avoid marshes, lakes, slow sandy streams, and large flat rivers, as well as dry steep ridges (Daugherty and Sheldon 1982). Adults are active terrestrially between May and September in western Montana; even during this period, movements may be restricted due to cold or dry weather (Daugherty and Sheldon 1982).

Forest cover appears to be an important element in tailed frog habitat (Metter 1964, Hawkins et al. 1988, Corn and Bury 1989, Welsh 1990, Bury et al. 1991). In the Mount Saint Helens vicinity (post-eruption), Hawkins et al. (1988) found that densities varied with forest condition, with low densities in nonforested stream basins, moderate numbers in forested areas, and the highest densities in basins with forested headwaters and nonforested areas downstream. They postulated that the high densities in the latter basin type may be due to suitable habitat upstream for adults and abundant food for tadpoles in the lower reaches, but noted that the fate of adults in the open reaches is uncertain due to their sensitivity to dessication. Changes to streams post-eruption were similar to those following clearcutting, including increased water temperature, high primary production by algae, and low inputs of terrestrial litter after the canopy is opened (Hawkins et al. 1988). In western Oregon, Corn and Bury (1989) found tailed frogs in 96% of unlogged areas and only 35% of logged areas; both density and biomass were higher in unlogged stands. Also in the Pacific Northwest, tailed frogs were caught in pitfall traps in closed-canopy forests, but were absent or rare in clearcuts (amounting to only 1% of the total catch, Bury et al. 1991). Welsh (1990:316) describes tailed frogs as "evolutionarily conservative elements of an ancient relictual forest ecosystem."

MODELING RULES:

- Select perennial stream lengths with mean gradient $\geq$ 5%.
- Buffer selected stream lengths by 30 m on either side.
- Use the buffer as a mask through which to filter cover types; within the mask, select for water and mature/overmature conifer types.
- Calculate the percentage of individual polygons predicted to contain habitat.

ASSUMPTIONS:

- Tailed frogs are most likely to use stream lengths with higher gradients; although a 5% cutoff is fairly conservative, the method for calculating gradient is not accurate enough to make narrow bounds practical. (Stream gradient is calculated by overlaying a raster file of perennial streams with a slope map constructed from a digital elevation model (DEM), then finding the mean slope for each stream length. Poor fits between the stream locations in the raster file and the stream beds on the DEM may yield inaccurate estimates of stream gradient.)
- Tailed frogs in the northern Rocky Mountains are not likely to venture much farther than 30 m from streambanks, although in more mesic regions of the Pacific Northwest, especially west of the Cascades, they may range more widely (see Metter 1967, Bury and Corn 1988).
- Mature/overmature conifer types are most likely to create the mesic microclimates important for tailed frogs.

## HARLEQUIN DUCK
*(Histrionicus histrionicus)*

HABITAT:

In western North America, harlequin ducks winter along the North Pacific coast, then migrate inland to nest along swiftly flowing mountain streams (Bellrose 1980). Studies of harlequin ducks in the northern Rocky Mountains have been conducted in Glacier National Park (Kuchel 1977), Grand Teton National Park (Wallen 1987), Flathead National Forest (Carlson 1990), and throughout Idaho (Cassirer and Groves 1991).

Harlequin ducks in Glacier National Park confined almost all activities to swiftly running waters (90% of area used), but also used abandoned meanders and other backwaters during periods of high water and as brood-rearing habitat (Kuchel 1977). In the early weeks of the brood-rearing period, females with broods used ponds more than running water. Females with broods also avoided all areas frequented by humans.

Four habitat characteristics were noted at more than 50% of harlequin duck observations in the Tetons (Wallen 1987): 1) streamside perennial shrub vegetation; 2) meandering (braided) channel types; 3) more than 3 loafing sites/10 m; and 4) areas unused by humans. Stream sections most suitable for harlequin breeding had gradients less than 1° and banks lined with dense perennial shrubs; breeding and brood-rearing occurred on streams with a mean gradient less than 3°. Wallen (1987) postulated that human activities may have a greater influence on breeding success than available habitat.

Harlequin ducks in Idaho primarily used riffle, run, and rapid streams with cobble/boulder substrate, second- to fifth-order, and over 50 m from roads (Cassirer and Groves 1991). Cassirer and Groves (1991) noted a difference in habitat use between ducks in North Idaho and those on the west slope of the Tetons (where habitat use was similar to that described by Wallen (1987)). Occupied streams in North Idaho were usually in mature/old-growth western red cedar/western hemlock or Engelmann spruce/subalpine fir stands. Cassirer and Groves (1991) suggested that the presence of mature/old-growth forest in North Idaho may indicate streams with high water quality, low sediment loads, intact riparian areas, and relative inaccessibility to humans -- important characteristics for harlequin ducks.

Further, it has been suggested that brood-rearing habitat may be limited on streams with low densities of harlequin ducks (Cassirer and Groves 1991). Streams with higher pair densities had characteristics of sites used by broods (narrower, meandering upstream reaches with lower gradient, smaller substrate, and more vegetative overhang, woody debris, and loafing sites). High-density streams in Idaho were also less accessible to humans and had a higher percentage of old growth as opposed to mature overstory.

Surveys for harlequin ducks were conducted on the Flathead National Forest, including Swan Lake Ranger District (Carlson 1990). Occupied streams were predominantly braided and canyon-type channels with gradients less than 2% and medium to high shrub or trees lining the banks. However, streams where harlequin ducks were found had the same characteristics as many streams where ducks were not found, suggesting that unmeasured parameters (like stream productivity and stability) may be limiting factors. No harlequin ducks were found on Swan Lake Ranger District, although Carlson (1990) conducted only limited surveys, and no historical records exist. Carlson (1990) recommended further survey of the district, especially stretches of the Swan River near Condon, but commented that most creeks on the district appeared to be small and to have extensive swampy areas.

MODELING RULES:

- Select all perennial streams.
- Calculate mean gradient for stream segments.
- Select stream segments with mean gradient less than 5%.
- Create a 30 m buffer around selected stream segments.
- Within the buffer, select for mature/overmature forest structure.
- Additionally, select for water, shrub, and broadleaf cover types.
- For individual polygons, calculate the percentage of area predicted to contain habitat.

ASSUMPTIONS:

- Due to similarities in habitat conditions, parameters identified for North Idaho can be appropriately applied to the Seeley-Swan study area. Parameters for the Tetons should be interpreted with greater caution with regard to the study area.

- The 5% cutoff for mean gradient is not inappropriately high. (Although the highest reported gradient was 3%, methods for calculating stream gradient are imprecise enough to justify inclusion of a slight buffer.)

- Failure to include accessibility of streams to humans will not invalidate the model; however, inclusion would certainly improve the model's accuracy.

- Other factors, such as stream stability and productivity (Carlson 1990), are not so important to harlequin ducks that the model will be invalidated without their inclusion.

# COMMON LOON
*(Gavia immer)*

## HABITAT:

Loons are thought to prefer large (> 4 ha) clear lakes with at least partially forested shorelines (Fitch 1989). Territories include an area with shallow water and emergent vegetation, and a secluded spot along the shore sheltered from waves. Loons tend to avoid water bodies with high human activity, large fluctuations in water level, turbid water, and completely barren shorelines.

## MODELING RULES:

- Select lakes at least 4 ha in size.
- Create 90 m buffer zones (shorelines) around selected lakes.
- Select for pole and mature/overmature forest stands within each buffer zone.
- Calculate the percentage of each buffer zone occupied by pole and mature/overmature forest stands.
- Select lakes with at least 25% of the buffer zone (not necessarily contiguous) in pole and mature/overmature forest stands.

## ASSUMPTIONS:

- The "partially forested" shorelines preferred by loons are adequately represented by a 25% cutoff.
- Heavy use by humans should not eliminate a lake from the analysis. The presence of loons at lakes with high levels of human activity indicates the ability of these lakes to support loons; such lakes are not necessarily sink habitat.
- The resultant predictions of loon distribution and habitat are not invalidated by an inability to include factors like water depth and turbidity in a GIS model.

# TOWNSEND'S WARBLER
*(Dendroica townsendi)*

## HABITAT:

Townsend's warblers are found in mature coniferous and mixed coniferous/deciduous forests. Their abundance is consistently higher in old-growth forests, and they exhibit sensitivity to fragmentation (Hejl and Woods 1991, Mannan and Meslow 1984, Tobalske et al. 1991). Dobkin (1992) recommends that Townsend's warbler be considered an interior-forest nesting species.

Mannan and Meslow (1984) provide the most complete description of nesting habitat. They located 15 nests in northeastern Oregon, all in old-growth stands, and generally on sites with high canopy volumes of grand fir and Douglas-fir. Mean canopy cover at nest sites was 63%. Grand fir and western larch were preferred foraging sites, while Douglas-fir and ponderosa pine were used less than expected. Mannan and Meslow (1984) attributed the low abundance of Townsend's warblers in managed stands to the near absence of grand fir, while the presence of this understory component in old-growth stands was credited with the higher abundances found there.

## MODELING RULES:

- Select mature/overmature forest structure.

- Select mixed conifer, Douglas-fir, western red cedar, and Engelmann spruce/subalpine fir cover types.

## ASSUMPTIONS:

- Although Dobkin (1992) labeled Townsend's warbler an interior-forest nesting species, R. Hutto and S. Hejl (pers. comm.) suggested against using this as a modeling criterion.

- Nesting habitat is of primary importance to Townsend's warblers; foraging conditions within suitable nesting habitat will be sufficient to support birds.

- Habitat conditions in northeastern Oregon are similar enough to northwestern Montana to allow extrapolation to the Seeley-Swan study area.

# BLACK-BACKED WOODPECKER
*(Picoides arcticus)*

## HABITAT:

The black-backed woodpecker has been described as a "species of denser forests, containing a diverse mix of species, no one of which is essential" (Bock and Bock 1974). While black-backed woodpeckers occupy spruce habitats, they are more frequently found in pines and other conifers typical of lower elevations and latitudes (Bock and Bock 1974). Weydemeyer and Weydemeyer (1928) noted that black-backed woodpeckers in Montana were most frequently found in mixed broadleaf/ conifer and Douglas-fir associations, and less commonly in ponderosa pine forests at low elevations and subalpine fir and lodgepole pine forests of the higher mountains. In North America, the northern limits of *Pinus* and the black-backed woodpecker are nearly identical (Bock and Bock 1974).

Black-backed woodpeckers respond opportunistically to insect outbreaks (Bock and Bock 1974, Lester 1980, Harris 1982). Lester (1980) studied woodpecker response to a mountain pine bettle epidemic in lodgepole pine forests of northwestern Montana, and found two black-backed woodpecker nests, both in dead lodgepole pine, 10-13 cm DBH. Harris (1982) examined post-fire responses of woodpeckers in the vicinity of Missoula, Montana, and observed a decline in woodpeckers three years after the fires occurred. Of the woodpeckers studied, black-backed woodpeckers nested in the smallest DBH trees (mean 23 cm) and the densest stands (1170 trees/ha). Western larch was used more than expected for both nesting and foraging.

Habitat use by black-backed woodpeckers during an insect outbreak was also studied by Bull et al. (1986). They found 15 black-backed woodpecker nests in northeastern Oregon: 67% in ponderosa pine, 27% in lodgepole pine, and 6% in western larch trees. Canopy closure averaged 46% at nest sites, and the mean DBH of nest trees was 37 cm. Nests were found in equal proportions in ponderosa pine, ponderosa pine/Douglas-fir, and grand fir stands. Live lodgepole pine was preferred for foraging, and was used 54% of the time. Black-backed woodpeckers foraged almost exclusively on ridges (97%).

MODELING RULES:

- Select for mixed conifer, Douglas-fir, ponderosa pine, lodgepole pine, and Engelmann spruce/subalpine fir cover types.
- Select for mature/overmature stands.
- In addition, select for recently burned cover types, mapping these as a separate habitat type.

ASSUMPTIONS:

- Areas defined as recently burned in the 1930s and 1990s vegetation files will still support high concentrations of insects, and thus be of value as black-backed woodpecker habitat.
- Recently burned areas will offer the highest quality habitat, but other selected stands will also be suitable.

# PILEATED WOODPECKER
*(Dryocopus pileatus)*

## HABITAT:

Forests with an old-growth component of western larch, ponderosa pine, or black cottonwood "seem to be essential for long-term support" of pileated woodpeckers in the northern Rocky Mountains (McClelland 1979). Based on work in northeastern Oregon, Bull (1987) described the best pileated woodpecker habitat as mature forest with at least 2 canopy layers; large live trees to provide cover (and to eventually replace dead trees); large dead trees for nesting; and standing dead trees and downed woody material for feeding.

Pileated woodpeckers in northwestern Montana are resident on feeding territories throughout the year (McClelland 1979), and may use 200-400 ha (McClelland et al. 1979). Bull (1987) also found that pileated woodpeckers used fairly large areas; based on the mean distance between nearest nests (1.48 km), each pair was allocated 220 ha.

At Coram Experimental Forest in northwestern Montana, McClelland et al. (1979) found that cavity nesters, including pileated woodpeckers, preferred western larch as a nest tree even though Douglas-fir was 5 times more abundant. Ponderosa pine was uncommon in this study area, but appeared to be a preferred nest tree; large black cottonwoods were also favored. Western larch and ponderosa pine have also been identified as preferred nest trees in northeastern Oregon (Bull and Meslow 1977, Bull et al. 1986, Bull 1987, Bull et al. 1992).

Strong selection for nest trees greater than 54 cm was observed by Bull (1987). McClelland et al. (1979) noted that pileated woodpeckers are unable to use trees much smaller than 20" due to their size, and that they prefer old-growth nest trees. Bull (1987) found 54% of nest sites in mature stands, 21% in old growth stands, and 24% in young stands with a few large trees. Canopy closure at pileated woodpecker nest sites averaged 74% (Bull et al. 1986), which corresponds with McClelland's (1979) description of favored nesting habitat as "dense forest."

Bull (1987) found 67% of nest sites in grand fir forest types, while the remainder were in mixed ponderosa pine and Douglas-fir. It should be noted that the dominant crown class at 80% of the nest sites was ponderosa pine or Douglas-fir, while the codominant crown class was grand fir or Douglas-fir at 85% of sites.

Topographic position of nest sites appears to be variable. Bull (1987) found 86% of nest trees on flat ground (ridges) or the mid-upper 33% of slopes; 68% of nests were on north or east aspects. Mean distance to water was 514 km (Bull et al. 1986). Others (see Bull 1987) have reported nesting near water and in valleys or bottomlands, and McClelland et al. (1979) describe optimal hole-nesting habitat as productive sites, especially wet bottomlands.

With regard to feeding habitat, Bull and Meslow (1977) noted that denser mixed-species stands were used more, and that Douglas-fir and western larch trees were preferred feeding sites. Grand fir types were found to contain 64% of feeding sites (Bull et al. 1986, Bull 1987).

MODELING RULES:

- Select for mature/overmature forest structure.
- Select for broadleaf, mixed conifer, Douglas-fir, ponderosa pine, western red cedar, and Engelmann spruce/subalpine fir cover types.

ASSUMPTIONS:

- Nesting and feeding habitat are similar enough to be treated as one block.
- Aspect is not an important factor in nest site selection, even though Bull (1987) recorded a majority of nests on north and east aspects. Similarly, the model will not be compromised by an inability to select stands based on density.
- Distance to water is not an important factor in the Seeley-Swan study area, where most stands would be within 0.5 km of water.
- Pileated woodpeckers require mature/old growth forest conditions. This assumption may be invalid for western Oregon (Mellen et al. 1992), where pileated woodpeckers have found to use immature stands. However, in the less-productive forests of the northern Rocky Mountains, pileated woodpeckers and mature/old growth forest stands appear to be connected (see above).

# FLAMMULATED OWL
## (*Otus flammeolus*)

HABITAT:

Flammulated owls are associated with mature to old-growth ponderosa pine and Douglas-fir forests throughout the northern and central Rocky Mountains (Bull and Anderson 1978, Goggans 1986, Holt and Hillis 1987, Howie and Ritcey 1987, Reynolds and Linkhart 1987, Atkinson and Atkinson 1990, Bull et al. 1990, Reynolds and Linkhart 1992). Stands used by flammulated owls also tend to be relatively open (Goggans 1986, Howie and Ritcey 1987, McCallum and Gehlbach 1988, Atkinson and Atkinson 1990). In Montana, all records of vocalizing or nesting flammulated owls are associated with old-growth ponderosa pine stands, although observations are not numerous (Holt and Hillis 1987).

In northeastern Oregon, stands of large-diameter (>50 cm DBH) ponderosa pine and Douglas-fir or grand fir with ponderosa pine in the overstory were identified as nesting habitat (Bull and Anderson 1978, Bull et al. 1990). Similarly, Goggans (1986) described nesting habitat as stands of ponderosa pine/Douglas-fir, 30-50 cm DBH, with less than 50% canopy closure. Flammulated owls foraged in the edge between forest and grassland, as well as in ponderosa pine/Douglas-fir forests of low or moderate density. Density appeared to be a crucial aspect of roosting habitat: flammulated owls roosted in mixed conifer stands, and avoided open stands of ponderosa pine.

Reynolds and Linkhart (1987, 1992) have found a strong association between flammulated owls and old-growth ponderosa pine/Douglas-fir habitat in Colorado, noting that such forests were used more than expected for nesting, foraging, and singing. They speculate that the presence of cavities and snags, the abundance of arthropods, and a stand structure suitable for foraging may be factors in this preference (Reynolds and Linkhart 1992). Flammulated owls have also been found to nest in live aspen ($n$ = 3) in Colorado (Richmond et al. 1980).

Marcot and Hill (1980) also recorded use of hardwoods; California black oak (*Quercus kelloggii*) was present at 67% of flammulated owl locations in northwestern California, while ponderosa pine was present at 50% of locations. All territory sites were on xeric midslopes or near ridgetops. Also in California, Bloom (1983) observed 3 flammulated owls, all in stands dominated by ponderosa pine.

In central Idaho, territorial flammulated owls occupied relatively open, multistoried Douglas-fir, ponderosa pine, and mixed conifer stands with some mature trees usually present (Atkinson and Atkinson 1990). Territories were often near more open areas, including old burns, grassy hillsides, natural clearings, or clearcuts. Atkinson and Atkinson (1990) also noted a clumped distribution of territorial males, leaving apparently suitable habitat vacant. Similarly, Marcot and Hill (1980) found "quasi-colonies" of territorial males, along with unoccupied areas of apparently optimal habitat.

At the northern edge of the flammulated owl's range in British Columbia, Howie and Ritcey (1987) identified mature/old-growth (> 100 year-old) Douglas-fir and Douglas-fir/ponderosa pine stands as nesting habitat, finding that owl densities were highest in stands 140-200+ years old. Stands were open, with canopy closures between 35-65%, and at least 2 canopy layers were present. Although Howie and Ritcey (1987:253) found a clearer association with mature/old-growth Douglas-fir than with ponderosa pine, they stated that "...the open nature of the fir forests coupled with natural or artificial openings created by logging probably resembles the physical structure of preferred forests in the southern portion of the owl's range."

MODELING RULES:

- Select ponderosa pine stands.
- Select mature/overmature forest structure.

ASSUMPTIONS:

- Any tendencies toward "quasi-coloniality" will not lead to gross overestimations of habitat occupied by flammulated owls.
- Given limited information on habitat use for Montana, characteristics described for northeastern Oregon, Colorado, and British Columbia can safely be extrapolated to the Seeley-Swan study area.
- As designed, the model is quite conservative. Because stand density cannot be incorporated in the selection process, including cover types that might be suitable at low stand densities (Douglas-fir and mixed conifer) would lead to an overestimation of available habitat. Thus, I opted to err in the opposite direction.

# BOREAL OWL
*(Aegolius funereus)*

## HABITAT:

Boreal owls are typically found in mature/old-growth spruce-fir forests in the northern and central Rocky Mountains (Palmer 1986, Hayward et al. 1987, Holt and Hillis 1987, O'Connell 1987, Ryder et al. 1987, Holt and Ermatinger 1989, Hayward et al. 1993). Although boreal owls may be relatively common in certain habitats, until recently they have remained little known in the Rocky Mountain states, probably due to their breeding chronology and high elevation associations (Holt and Hillis 1987).

Based on limited surveys in Montana, Holt and Hillis (1987) noted a preference for mature/overmature Engelmann spruce/subalpine forests below 1500 m. Holt and Ermatinger (1989) recorded the first confirmed nest in Montana, also in an Engelmann spruce/subalpine fir stand over 120 years old.

An extensive description of habitat use by boreal owls in the northern Rocky Mountains is provided by Hayward et al. (1993). Forests in Montana, Idaho, and northern Wyoming were surveyed for boreal owls, and 49 nests or singing males were found. No owls were detected below 1292 m, and 75% of locations were above 1584 m. Forest cover types in which owls were located included lodgepole pine, Douglas-fir, western hemlock/western larch/subalpine fir, Engelmann spruce, and Engelmann spruce/subalpine fir. Stands were classified as mature or older at 76% of locations.

Hayward et al. (1993) also studied boreal owls more intensively in the River of No Return Wilderness (RNRW) in central Idaho. Of 28 breeding sites in RNRW, 39% were in mixed conifer, 25% in Engelmann spruce/subalpine fir, 18% in Douglas-fir, and 18% in aspen stands. Lodgepole pine was not used for nesting, although it was the most common forest type in the study area. Nest sites were found only in structurally complex mature and old forests; characteristic features included high density of large trees, open understory, and multilayered canopy. The mean size of nest stands was 7.6 ha (range 0.8-14.6 ha). Average roost-to-nest distance was approximately 1730 m; Hayward et al. (1993) suggested that this measure is probably a good approximation of foraging distance. The best foraging habitat was associated with Engelmann spruce/subalpine fir stands, where prey densities were highest and open stand structure facilitated hunting. Mature/overmature Engelmann spruce/subalpine fir stands also provided cool sites for summer roosting, and Hayward et al. (1993) proposed that roosting habitat can be maintained through management of foraging habitat. Finally, Hayward et al (1993) further recommended that all forested sites within the spruce-fir zone be considered as potential boreal owl habitat, as well as forests 100-200 m below this zone, which may provide the most important nesting habitats.

MODELING RULES:

- Select mature/overmature forest size classes.
- From the above set, select mixed conifer, Engelmann spruce/subalpine fir, Douglas-fir, and broadleaf cover types.
- Finally, select mean elevations greater than or equal to 1300 m.

ASSUMPTIONS:

- Broadleaf cover types at high elevations will most likely be aspen, which was used for nesting in central Idaho (Hayward et al. 1993).
- Since mean elevations for stands are used in the modeling process, and some stands are quite large, the lower elevation limit is set conservatively at 1300 m, closely corresponding to the lowest elevation at which boreal owls were detected (Hayward et al. 1993).
- The mean size of nesting stands is smaller than the minimum mapping unit for the 1930s vegetation layer; therefore, no minimum stand size need be applied.
- Foraging and roosting habitat will be accounted for in the selection of nesting habitat, since mature/overmature Engelmann spruce/subalpine fir is included as nesting habitat. As a result, distances between patches need not be considered.

# BARRED OWL
*(Strix varia)*

## HABITAT:

The barred owl is widely distributed throughout North America, and has recently expanded its range into the western part of the continent, including western Montana (Shea 1974, Taylor and Forsman 1976, Boxall and Stepney 1982). Habitats used most frequently by barred owls include deciduous, coniferous, and mixed stands (Nicholls and Warner 1972, Shea 1974, Taylor and Forsman 1976, Leder and Walters 1980, Boxall and Stepney 1982, Holt and Hillis 1987, Dunbar et al. 1991). Barred owls require extensive areas of forest containing large mature and decadent trees to provide security and nesting cavities (Allen 1987); however, use of younger stands has also been observed (Leder and Walters 1980, Dunbar et al. 1991).

In Montana, nests have been located in mixed old-growth stands, typically in association with western larch (Holt and Hillis 1987). Barred owls have also been heard during the breeding season in riparian and lodgepole pine habitats at elevations of 900-1800 m (Holt and Hillis 1987) and in mixed Douglas-fir/lodgepole pine/western larch forests in the vicinity of Glacier National Park (Shea 1974). Breeding has occasionally been reported in the Blackfoot, Bitterroot, and Fisher River valleys (Flath, cited in Ellis et al. 1987).

In east-central Minnesota, Nicholls and Warner (1972) identified a preference for oak woods and mixed hardwood/coniferous forests, noting that these habitats seemed to provide all items essential for survival of barred owls, including hollow trees for nesting and a sparse understory to facilitate hunting.

Boxall and Stepney (1982) reviewed records of barred owls in Alberta between 1912-1980, finding that records post-1959 have been most concentrated in mixed-wood boreal forest as well as coniferous and montane forests. They suggested that barred owls in Alberta once preferred mixed-wood boreal forest (in correspondence with their preference for deciduous or mixed transitional forests in eastern North America), but have recently adapted to forests of predominantly coniferous composition. They also related the increased sightings in western Alberta to the recent appearance of barred owls in northern Idaho and western Montana, suggesting both a range expansion and an increase in numbers.

In southwestern British Columbia, an area of recent range expansion, Dunbar et al. (1991) observed barred owls most commonly in mixed stands of hardwoods and conifers in broad riparian corridors along major rivers and lakes; they also found a number of owls in upland stands of mature and old-growth coniferous forest. Many areas where barred owls are now most common were logged in the early 1900s. Their survey results demonstrate that barred owls have successfully colonized a broad range of habitats in British Columbia, as indeed seems to be the case throughout the species' range.

MODELING RULES:

- Select mature/overmature forest structure.
- Select broadleaf, mixed conifer, Douglas-fir, ponderosa pine, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir cover types.
- Select for elevations $\leq 1800$ m.

ASSUMPTIONS:

- Barred owls are generalists, able to use all forest cover types defined for this study.
- Barred owls are not exclusively tied to riparian areas; although such areas are frequently used, their distribution does not limit the distribution of barred owls.

## NORTHERN GOSHAWK
*(Accipiter gentilis)*

HABITAT:

Northern goshawks are most commonly found in dense, mature/old-growth stands (Reynolds et al. 1982, Crocker-Bedford and Chaney 1986, McCarthy et al. 1987, Hayward and Escano 1989, Whitford 1991). In northwestern Montana, northern goshawks typically nested in mature/overmature forest with a closed canopy (75-85%) on moderate slopes (15-35%) with northerly aspects (Hayward and Escano 1989). Nest sites were often located on lower slope positions, and in one of the older stands in an area. Both water and a large opening were usually within 0.5 km of nests. In Glacier National Park, a goshawk nest was located in the spruce-fir zone at 4500' (Parratt 1959). Whitford (1991) examined 12 nests on the Lewis and Clark National Forest in Montana: All nests were on north aspects; mean canopy closure was 72%; mean live tree DBH was 31 cm; and mean live tree age was approximately 200 years.

Goshawk habitat has also been studied in northeastern Oregon, broadly defined to include the west slope of the Cascades, by Reynolds et al. (1982). Goshawks there were found to nest on gentle slopes with northwest to northeast aspects in dense, mature conifer stands. A majority of nests were located in old-growth stands, and mixed conifer, fir, and pine cover types were used. Canopy closure averaged 60%. About two-thirds of the nests were less than 0.5 km from water, but based on the locations of the remaining nests, water does not appear to be a requirement. In general, shaded, mild environments and protected sites were used. Moore and Henry (1983) also examined goshawk nest sites in northeastern Oregon. Stands of larger conifers (mean DBH 22.1 cm) with relatively low understory crown volume were used. Douglas-fir and western larch were preferred nest trees, and a majority of nests were located on north or flat aspects.

On the east slopes of the Sierra Nevada, the goshawk is considered an ecological indicator of mature/old-growth forests (McCarthy et al. 1987). In a habitat model developed for northeastern California (Shimimoto and Airola 1981, in McCarthy et al. 1987), stands with the following characteristics were considered suitable habitat: red fir, lodgepole pine, Jeffrey pine, aspen, and mixed conifer communities; >40% canopy cover of trees with >28 cm DBH; a minimum size of 12 ha; at least 25% stand canopy; 0-30% slope; and less than 1.7 km to water.

In northern Arizona, Crocker-Bedford and Chaney (1986) found that dense stands provided better goshawk habitat; good nest stands had at least 79% canopy cover, while marginal stands had at least 60% canopy cover. The vast majority of the canopy came from trees >25.4 cm DBH, and nest stands had much higher densities of large trees than typical stands within the study area. Ponderosa pine stands were more likely to be on north aspects than mixed conifer stands; Crocker-

Bedford and Chaney (1986) speculated that dense canopies in mixed conifer stands may be enough to provide a cool microclimate. Only 8 of 43 nests were within 1 km of water. Crocker-Bedford and Chaney (1986) recommended that nest stands should be at least 8 ha, and that fully suitable nesting habitat required at least 2 alternate stands less than 1 km apart.

MODELING RULES:

- Select for mature/overmature forest structure.

- Select mixed conifer, Douglas-fir, ponderosa pine, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir cover types.

- Select slopes less than or equal to 40%.

- Select stands with ≥50% area in northerly aspects (less than or equal to 45°, or greater than or equal to 315°).

ASSUMPTIONS:

- Nest sites are the most critical aspect of goshawk habitat, since researchers focused only on nesting without discussing foraging and roosting habitat.

- Distance to water is not a limiting factor in goshawk nest site selection.
- Stand density appears to be an important factor in nest site selection. Although density could not be included, the model already seems overly restrictive, and I would assume that stands meeting the modeling criteria above would to a great extent provide the cool, shaded microclimate most likely to be found in dense stands.

# BALD EAGLE
*(Haliaeetus leucocephalus)*

## HABITAT:

The *Habitat Management Guide for Bald Eagles in Northwestern Montana* (MBEWG 1991) provides an excellent overview of habitat characteristics, and was a primary source used in determining modeling rules. In particular, this guide contains summary tables outlining attributes of bald eagle habitat and refining these attributes into easily-adaptable rules for modeling habitat using a GIS.

In selecting nesting habitat, bald eagles usually prefer late-successional forests in close proximity to water and with relative isolation from human disturbance (MBEWG 1991). In northwestern Montana (Zone 7 for bald eagle management and recovery), all nest sites were within 1 mile of a lake or reservoir larger than 40 acres or a stream greater than fourth order in size (Wright and Escano 1986), denoting the importance of proximity to an adequate prey base.

Nesting stands and nest trees are selected based on structure. Multi-layered mature/old-growth forests are strongly preferred (Grubb 1980, Anthony et al. 1982, USFWS 1986, Wright and Escano 1986, Stalmaster 1987, Anthony and Isaacs 1989). Often, more than one nest site is available within selected stands. Nest trees are typically mature or overmature with open crowns and sturdy limbs, and occupy dominant positions within stands (Grubb 1980, Wright and Escano 1986, Stalmaster 1987, Anthony and Isaacs 1989, Jensen 1988). Ponderosa pine, Douglas-fir, and cottonwood trees are most frequently selected in western Montana (Wright and Escano 1986), probably because their typical growth forms are able to support large nests (MBEWG 1991). Nest position in relation to associated water bodies is an important factor: In western Montana, all nests were within topographic line-of-sight of water; all were ≤450' in elevation above the associated water body; and in 90% of cases, nests were ≤2000' in distance from the associated water body (Wright and Escano 1986).

Lakes, reservoirs, rivers, and open upland areas provide foraging habitat for bald eagles (MBEWG 1991). Perch sites are an important attribute of foraging habitat (Fielder and Starkey 1986); proximity to potential prey, isolation from disturbance, good visibility of the surrounding landscape, and accessibility for landing and departure are critical components of preferred perches (Stalmaster 1987). Similarly, perch sites, roost sites, and prey availability are the essential elements of winter habitat.

MODELING RULES:

- Select major rivers manually, including the Swan River downstream from Lindbergh Lake, and the Clearwater River downstream from Rainy Lake. Also select lakes $\geq 16$ ha (40 acres) in size.

- Create a 1610 m (1 mi) buffer on each side of selected rivers and lakes.

- Eliminate polygons that are completely outside of this buffer zone. For polygons falling at least partly within the buffer:
  - Select mature/overmature deciduous, mixed conifer, Douglas-fir, and ponderosa pine forest types, and water.

  - Select slope $\leq 40\%$.

  - Select elevations $\leq 1385$ m.

ASSUMPTIONS:

- The limiting factor for bald eagles appears to be nesting habitat near open water for foraging; as designed, the model should account for both summer and winter habitat. However, some factors which may be critical in determining habitat use were not included in the model, such as stand size, elevation above water body, line-of-sight position with regard to water body, distance to late winter food source, and distance from open road. Stand size was not included because I felt the model was already fairly restrictive; it could easily be included in further analyses. Elevation above water body and line-of-sight position are difficult to model using a GIS, and were thus excluded due to time constraints. Data on late winter food sources and open roads were not available for the entire study area.

- Lacking an ability to select rivers based on stream order, a conservative approach to manual stream selection was taken. Additional stream lengths, especially in the vicinity of Holland and Lindbergh Lakes, may be suitable as well. However, the sections I selected were approved upon preliminary review by the Montana Bald Eagle Working Group (December 1993).

- The elevational cutoff employed may lead to elimination of some available habitat, but it seems more likely that inclusion of less-productive lakes at higher elevations would overestimate suitable habitat. The 1385 m cutoff was recommended by Bill Ruediger (pers. comm.) of the Montana Bald Eagle Working Group.

# PEREGRINE FALCON
(*Falco peregrinus anatum*)

## HABITAT:

The peregrine falcon (*anatum* subspecies) has been on the Endangered Species list since 1972. A recovery plan for the Rocky Mountain/Southwest population was approved in 1984; unless otherwise cited, all information below was taken from the recovery plan. Essential habitat was delineated on National Forest lands in the Northern Region in 1978; none fell within the study area boundary (USDA:FS 1978). However, a pair of peregrine falcons was documented in the Swan Valley during the 1993 nesting season, and currently there are one or two nesting pairs (USDA:FS 1994a).

Peregrine falcons prefer to nest on cliffs, or series of cliffs, that tend to dominate the surrounding landscape; mountain valleys and river gorges with steep cliffs are also preferred (USDI:FWS 1984). For the Rocky Mountain/Southwest population, remaining occupied eyries are on cliffs usually 200-300' high (range 40-2100'). Most nests are <9500'; nesting is rare above 8500'. Preference for southern exposures increases with latitude. Nest sites are often adjacent to water, and the majority of eyries in the Rocky Mountain/Southwest Region are within one mile of a stream or river.

Peregrine falcons may travel up to 17 miles from cliffs to hunting areas; normally, an adequate food source is to be found within 10 miles of an eyrie. Preferred hunting habitats, because of the abundance of avian prey to be found, include cropland, meadows, river bottoms, marshes, and lakes.

## MODELING RULES:

- Select cliff habitats for nesting: areas with slope ≥150% (based on a slope map generated from the digital elevation model), 2 ha or larger (to exclude very small outcrops from the analysis), below 2615 m (8500'), and with at least a 90 m change in elevation within each polygon (to approximate cliff height).

- Calculate euclidean distances between cliff habitats and all lakes and streams. Select only cliff habitats within 1610 m (1 mi) of water.

- Select water, grass, and agriculture as foraging areas.

- Again based on euclidean distances, select only foraging areas within 16,100 m (10 mi) of cliff habitats.

- For nesting habitat, calculate the percentage of each individual polygon predicted to be occupied by cliffs.

ASSUMPTIONS:

- Nesting habitat is assumed to be the limiting factor for peregrine falcons, but foraging habitat is modeled as well because an adequate prey base is necessary to support an eyrie.

- $\geq$150% slope is an adequate indicator of cliff habitat.

- The cover types listed above provide the best hunting for peregrines. Shrub types could have been included as well, but, because of their confusion with clearcuts in the classification process, may have yielded an overestimate of foraging habitat. As it is, foraging areas are most likely underestimated.

# MARTEN
*(Martes americana)*

## HABITAT:

Optimal habitat for the marten has been described as mature/old-growth spruce-fir forest with at least 30% canopy cover, plentiful fallen logs and stumps, and a lush understory of shrubs and forbs (Burnett 1981, Clark and Casey 1989); mixed coniferous and deciduous forests are also used (Clark and Casey 1989). "Old-growth spruce-fir forests in the western United States provide much of the remaining marten habitat." (Burnett 1981:95) In Glacier National Park, Burnett (1981) found that adult marten were concentrated in mesic spruce and larch cover types, although a variety of cover types were used. Marten preferred stands with canopy cover >17% (mean 35%). Small mammal densities were highest in mesic spruce cover types.

Voles, a common item in marten diets, were also most abundant in mesic habitats in the Selway-Bitterroot (Koehler et al. 1975, Koehler and Hornocker 1977). Marten were found to use a variety of forest types in winter, but activity was highest in Engelmann spruce/subalpine fir stands with mesic habitat types, >30% canopy cover, and overstory age >100 years. Similarly, in north central Washington, marten frequented older (≥82 years) spruce-fir and lodgepole pine forests in winter; voles and red squirrel middens were available there (Koehler et al. 1990) Presumably, marten activity is highest in mature forests because of their associated abundant vole populations (Koehler et al 1975). In the Selway-Bitterroot in winter, marten crossed openings up to 300' wide, but did not appear to hunt in such areas; marten were not seen to cross openings >300' in width. However, open areas, meadows, burns, and other habitats avoided by marten in winter may be used in summer and fall if they offer adequate food and cover (Koehler et al. 1975, Koehler and Hornocker 1977).

In the northern Sierra Nevada, marten were found to prefer areas within 60 m of meadows and rarely used sites >400 m from meadows; however, marten avoided open areas in all seasons (Spencer et al. 1983). As in other areas, marten selected for tall, dense forests with many large snags, stumps, and logs.

Marten rested primarily in subnivean sites associated with coarse woody debris, including logs and stumps, in southeastern Wyoming (Buskirk et al. 1989). Spruce-fir stands received more use than expected by adults, whereas lodgepole pine was used less than expected based on availability. Spruce-fir stands contained 75% of the resting sites associated with coarse woody debris. Resting sites were also closer than expected to lakes and streams (mean 173 m). The apparent dependence of marten on old-growth forests in the central Rockies in winter may be explained by the importance of resting where coarse woody debris is available to provide thermal cover (Buskirk et al. 1989).

In addition to winter thermoregulatory needs, a preference for dense, mature coniferous forest or mixed forest may also be explained by overhead cover from predation, and prey abundance and availability (Bissonette et al. 1991). In landscapes altered by timber harvest, residual forest patches ≥15 ha with shapes tending toward unity would seem to be desirable elements for marten; in such landscapes, old growth should be the matrix element (Bissonnette et al. 1991). In an extensively harvested landscape in western Newfoundland, almost 90% of marten captures were in residual forest stands; data from this study clearly demonstrate that larger residual and undisturbed stands (>15 ha) are important habitat components (Snyder and Bissonette 1987). In north central Maine, marten densities were lower and home range lengths greater in clear-cut than in undisturbed or partially harvested forest (Soutiere 1979).

In summary, Koehler et al. (1975) state that, in the northern Rocky Mountains, marten prefer high-elevation basins dominated by spruce and subalpine fir or mountain hemlock. They note that alpine forests (whitebark pine stands, for example) offer good habitat as well, particularly when adjacent to dense, mature forests at lower elevations. Mature lodgepole pine is also suitable in moist habitat types, and in areas of high precipitation, dense cedar-grand fir forests at lower elevations provide habitat for the marten as well. Koehler et al. (1975) further suggest that dry stands of ponderosa pine, inland Douglas-fir and associated species will rarely be used by marten except as travel routes.

<u>MODELING RULES</u>:

- Select mature/overmature mixed conifer, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir cover types.
- Select stands ≥15 ha.

<u>ASSUMPTIONS</u>:

- Winter habitat, important for cover, foraging, and subnivean rest sites, seems to be the limiting factor for marten. Presumably, winter habitat will receive the most use throughout the year for foraging and denning, although younger stands and open areas may receive some foraging use in summer.

- Selection for high-density stands would have improved the model; as it is, habitat may have been overestimated. However, some cover types excluded based on recommendations by Koehler et al. (1975), like Douglas-fir, may actually be suitable, thus leading to underestimation.

- Road densities were not included in the model; although high road densities are expected to increase trapping vulnerability, I was unable to find specific research to document this trend.

## FISHER
*(Martes pennanti)*

HABITAT:

Fisher ecology and management in the western United States was reviewed by Heinemeyer and Jones (1994); they provide the following overview of habitat characteristics. In the West, fishers are usually found in coniferous forests including diverse habitat types and successional stages. Close association with forested riparian areas has been noted; these areas are used for foraging, resting, and travel. Although fishers use a variety of successional stages, most western studies have identified a preference for mature/old-growth forests. Avoidance of openings may be somewhat dependent on season and vegetation; clearings may be used if a shrub layer is present to provide cover. Most studies have suggested that fishers are tolerant of moderate levels of human activity, but populations may be indirectly affected by removal or fragmentation of habitat and increased trapping accessibility. Fisher populations declined in the early 1900s -- most likely because of habitat lost through settlement and logging, overtrapping, and predator poisoning -- and western populations remain at low levels. Fishers were reintroduced to the Swan Valley in 1959-60, when 15 individuals were released near Holland Lake (Weckwerth and Wright 1968).

In north-central Idaho, most fisher observations were in mesic grand fir habitat types (Jones 1991). Grand fir and Engelmann spruce dominated stands used in summer; similarly, in winter, grand fir, Engelmann spruce, and lodgepole pine dominated stands. Summer habitat had a relatively high component of moderate to large DBH Engelmann spruce, large DBH Douglas-fir, and pacific yew; stands with a strong lodgepole or ponderosa pine component were avoided. Winter habitat included stands with a relatively high basal area in Douglas-fir and lodgepole pine. On averages, home ranges contained 53% mature/old-growth stands. In the summer, 90% of observations were in mature/old-growth forest; in the winter, 54% were in mature/old-growth and 46% in young forest (Jones and Garton 1994). Mature/old-growth stands were used extensively for resting, while hunting occurred in a range of successional stages. Stands with canopy cover >60% were preferred for resting and >80% for hunting. Fishers strongly selected wetland forest types, with selection for forested riparian habitats evident at several scales in summer and winter (Jones 1991). In summer, 50% and 75% of observations were withn 15 and 23 m of water. In moving across landscapes, fishers commonly used forested riparian areas, where preferred resting habitat and prey may be more available than in surrounding habitats.

Reintroduced fishers in the Cabinet Mountains of northwestern Montana preferred mixed conifer and cedar/hemlock stands, avoiding subalpine fir and hardwood (typically alder or recently cut) habitats (Roy 1991). Dense stands of young (pole) to moderately-aged forest were preferred. As the fishers established permanent home ranges, they used predominantly mesic forested habitats, often mixed stands of grand fir, cedar, and hemlock (Heinemeyer 1993). They also showed preference for low-elevation, low-gradient, north-facing areas near water. Areas >1200 m were avoided. Fishers selected areas near perennial streams, rivers, marshes, and lakes. Sixty-five percent of fisher locations were within 200 m of water, and areas <400 m from perennial streams were selected by fishers.

MODELING RULES:

- Create a 400 m buffer around all perennial streams, lakes, and wet meadows.

- Filter selected vegetation through the buffer, and include only the portions of polygons falling within the buffer:
  - Select mature/overmature mixed conifer, Douglas-fir, lodgepole pine, western red cedar, and Engelmann spruce/subalpine fir cover types as yearround habitat.
  - Select pole stands of the same cover types as winter habitat.
- For individual polygons, calculate percentage of area predicted to be habitat.

ASSUMPTIONS:

- The limiting factor for fishers appears to be summer habitat, because use is most restricted at that time. Winter habitat was also modeled, however, to provide additional information on habitat conditions.

- A 400 m buffer around hydrographic features is a fairly generous estimate of high-quality habitat; in the Cabinets, selection for areas within 200 m of water was significant throughout the year (Heinemeyer 1993), and in north-central Idaho, most locations were in even closer proximity to water (Jones 1991).

- An elevational cutoff was not employed, despite Heinemeyer's (1993) findings that fishers avoided areas >1200 m. I assumed that the juxtaposition of selected cover types with perennial water and lakes would be sufficient to exclude unsuitable areas.

- Road densities were not included in the model, although they may indicate relative vulnerability to trapping. The spatial correspondence between roads and riparian areas within the Seeley-Swan suggests that most habitat would be found unsuitable if road densities were considered.

- The model would also have been improved by exclusion of low density stands.

# WOLVERINE
*(Gulo gulo)*

HABITAT:

Wolverines were studied by Hornocker and Hash (1981) in the South Fork of the Flathead River drainage, one drainage east of the Swan Valley. Wolverines seemed to select *Abies* cover types throughout the year, but especially in summer; 56% of locations were in *Abies* types when all seasons were averaged. Wolverines were most frequently found in large areas of medium or scattered mature timber (70% of locations). Remaining locations were in ecotonal areas, small timber pockets, and rocky, broken areas of timbered benches. Wolverines were rarely located in burned-over areas or wet meadows, and dense young timber received the least use. However, seral lodgepole pine and western larch sites were frequently used. Hornocker and Hash (1981) suggested that food availability is the main factor determining movements and range of wolverines in the South Fork drainage. Carrion and prey items were apparently more available in mature or intermediate stands preferred by wolverines, particularly the edge and ecotonal areas around cliffs, slides, blowdowns, basins, swamps, and meadows.

Wolverines appeared to meander through timber, and straight-line movements across large openings also were observed. No wolverines were relocated in a clearcut of any size, but tracks were seen to cross clearcuts 15 times. Wolverines were located within 1-3 km of clearcuts and active roads 12 times; males were found farther from clearcuts, roads, and burns than females.

Lower elevations were used more in winter than in summer (means of 1371 m and 1920 m respectively). Areas of all exposures were used, but easterly and southerly exposures were used more consistently. Use of a variety of topographic positions was noted, including slopes (36% of locations), basins (22%), wide river bottoms (14%), and ridgetops (8%).

No differences in wolverine density, movement, habitat use, or behavior were found between the managed and wilderness portions of the South Fork drainage. Hornocker and Hash (1981) postulated an effective separation of humans and wolverines due to limited human access in winter and use of higher elevations by wolverines in summer. However, 15 of 18 known mortalities between 1972-1977 were caused by humans.

Hornocker and Hash (1981) found that individual wolverines ranged widely; average yearly ranges were 422 $km^2$ for males and 388 $km^2$ for females. Sizeable home ranges require equally sizeable scales for analysis: "Relative to other species in northwestern Montana, the wolverine population must be treated as regional rather than local." (Hornocker and Hash 1981:1293) With regard to wolverine management, Hornocker and Hash (1981) recommended leaving basins, southerly and easterly slopes, and edge/ecotonal areas intact.

MODELING RULES:

- Select mixed conifer, Douglas-fir, lodgepole pine, and Engelmann spruce/subalpine fir cover types.

- Select pole and mature/overmature stands.

- Additionally select for whitebark pine/Engelmann spruce/ subalpine fir cover type.

- Filter out polygons below 1300 m.

- Select for easterly and southerly aspects (45-225°).

- Buffer selected stands by 2 pixels (60 m).

- Identify and include buffered sections that overlap with the following cover types: barren, grass, shrub, and rocky, scattered trees.

- For areas of overlap identified above, find the percentage of each polygon included in the overlap area so that accurate estimates of the area of predicted habitat can be obtained.

ASSUMPTIONS:

- Because a variety of topographic features are used by wolverines, it is not necessary to select for specific features in a habitat model.

- The method outlined above is an adequate and accurate means of identifying ecotonal areas.

- No minimum stand size need be identified.

- Inclusion of road density (as an indicator of security) is not a necessary factor, especially given the 1300 m elevation cutoff, which eliminates many roaded areas.

# LYNX
## (*Felis lynx*)

HABITAT:

Although habitat selection by lynx is not well understood, presumably, good habitat for the snowshoe hare (*Lepus americanus*), the lynx's primary prey, is also good habitat for lynx (Quinn and Parker 1987). Lynx, however, need a mosaic of forest types, including early successional stages for hunting and mature forests for denning (Koehler and Brittell 1990).

In north-central Washington, snowshoe hares were most abundant in 20-year-old lodgepole pine stands (Koehler 1990). Lynx used lodgepole pine and Engelmann spruce/subalpine fir forest types more than expected based on availability in this area; xeric lowland forest types were used less than expected. Lynx used areas above 1463 m, and were located at higher elevations in summer than in winter. Four denning sites were located in mature ($\geq$250 years old) stands with Engelmann spruce, subalpine fir, and lodgepole pine in the overstory; all were on north-northeast aspects and had numerous downed logs.

In northwestern Montana, most relocations for two radio-collared lynx were in young, dense lodgepole pine: 90% were in stands generated following the 1910 fires, and the rest were in mature Douglas-fir/western larch riparian stringers within the burn (Koehler et al. 1979). Xeric sites where lodgepole pine was dominant contained 88% of the locations in the 1910 burn; the other 3 locations were in mesic sites dominated by subalpine fir and Engelmann spruce. Snowshoe hares were most abundant in densely-stocked stands of lodgepole pine (<80 years old).

MODELING RULES:

- Select pole-sized lodgepole pine cover types as foraging habitat.
- Select mature/overmature Engelmann spruce/subalpine fir or lodgepole pine cover types as denning habitat.
- As an additional criterion for denning habitat, select stands where 50% or more of the area falls on north-northeast aspects (315-360° and 0-135°). Majority aspect was used instead of mean aspect because means are highly inaccurate for northerly aspects.

ASSUMPTIONS:

- Pole-sized lodgepole is a fair representation of foraging habitat for lynx. This assumption is tenable: In both Koehler et al. (1979) and Koehler (1990), lodgepole pine stands presumably pole-sized by their ages received use by snowshoe hares). However, the model would be improved by the inclusion of sapling lodgepole pine stands, which was not possible. All seedling/sapling stands would have had to be included for results to be comparable for the two time periods, and this would have probably greatly overestimated foraging habitat. As it is, the model most likely underestimates foraging habitat.

- Although snowshoe hares prefer high-density stands (see Koehler and Brittell 1990), density was not included in the model because it was not comparable for the two time periods.

- Travel corridors are assumed not to be a limiting factor for lynx in the Seeley-Swan based on visual examination of vegetation maps.

# GRAY WOLF
## (*Canis lupus*)

HABITAT:

Wolves have had a place on both Montana and federal endangered species lists since 1973. A recovery plan for the northern Rocky Mountains was completed in 1980, then revised and approved in 1987; unless otherwise cited, all information summarized below was taken from the recovery plan (USDI:FWS 1987).

Historically, wolves have used a broad spectrum of habitat types, occupying nearly all habitats in the northern hemisphere except for true deserts. Use of a variety of habitat types is likely related to the large areas used by wolves: Pack territories normally range from 50-200 $mi^2$, but may encompass thousands of miles. Key components of wolf habitat include: 1) a sufficient year-round prey base of ungulates and alternate prey; 2) suitable and somewhat secluded denning and rendezvous sites; and 3) sufficient space with minimal exposure to humans. Ungulate prey species include elk, mule deer, moose, white-tailed deer, and mountain goat; in the Rocky Mountains, primary prey are elk, moose, and deer. Beaver and smaller mammals serve as supplemental prey. Areas important to wolves include ungulate summer and winter ranges, calving and fawning areas, and riparian habitat.

Wolves are especially sensitive to human disturbance at denning and initial rendezvous sites. Dens are commonly on southerly aspects of moderately steep slopes, usually within 400 yards of surface water and at an elevation overlooking low-lying areas. One den in Glacier National Park consisted of openings dug into a flat-topped, heavily forested knoll next to a 2 ha meadow (Ream et al. 1989). Rendezvous sites are typically complexes of meadows and adjacent timbered slopes, also with surface water nearby.

Minimal exposure to humans can be critical to wolf survival. In the North Fork of the Flathead River drainage, at least 13 of 14 known mortalities were human-caused (Pletscher et al. 1991). As road densities increase, so does human access; thus, high road densities may threaten wolf populations (Mech et al. 1988). An examination of percent mortality by region in Minnesota showed an inverse relationship between human density, road density, and viable wolf populations (Thiel 1985). Also, from 1926-1960 in Wisconsin, wolves failed to survive in counties with mean road densities $>0.93$ mi/$mi^2$ (Thiel 1985). Mech et al. (1988) showed that areas in Minnesota had road densities below the threshold listed by Thiel (1985), and unoccupied areas were well above the threshold. More recent analyses may show that road densities play a lesser role in persistence of wolf populations than previously thought (Pletscher, pers. comm.); nonetheless, the density of roads in an area certainly influences its suitability for wolves.

MODELING RULES:

- Select vegetation polygons falling within winter ranges for elk, moose, white-tailed deer, mule deer, and mountain goats, as delineated by Montana Fish, Wildlife, and Parks (FWP) biologists (layer acquired May 1992, Bissell, pers. comm.).

- Select vegetation polygons falling within areas with mean road densities $<3$ mi/mi$^2$ (Pletscher, pers. comm.); include all roads in the density calculation regardless of closure status.

ASSUMPTIONS:

- An adequate prey base is the most critical factor in sustaining wolf populations.

- Ungulate winter ranges are probably more important to wolves than summer ranges; however, summer ranges would have been included in the model if they had been mapped for each species by FWP biologists. Summer range was mapped for mountain goats, but was not included in the model for 3 reasons: 1) mountain goats are not a preferred prey item in the Rocky Mountains; 2) mountain goats use higher elevations in summer, and wolves prefer lower elevations (Ream et al. 1985); and 3) consistency would be lost if summer range were included for one species only.

- Because contact with humans is highly probable, areas with total road densities $>3$ mi/mi$^2$ will likely be sinks for wolves and should be excluded from maps of suitable habitat. The threshold value selected is probably a generous one, and could easily be lowered, perhaps to $>1$ mi/mi$^2$. It also might be better to base cutoff values on open road densities rather than including all roads, open and closed, in density calculations.

- Lacking comprehensive data on roads in the 1930s, it is assumed that no areas had road densities $>3$ mi/mi$^2$ at that time.

## GRIZZLY BEAR
(*Ursus arctos*)

HABITAT:

The grizzly bear may be the most logical example of an umbrella species in the northern Rocky Mountains because of its large home range and the variety of habitats it occupies (USDI:FWS 1993). Prime grizzly habitat is characterized by diversity; to ensure a varied food supply, a wide range of vegetative types is essential (USDI:FWS 1993). In addition, security from human disturbance is an important aspect of optimum habitat (see Brannon et al. 1988; McLellan and Shackleton 1988, 1989; Mattson et al. 1992; Mace and Manley 1993; USDI:FWS 1993).

In the Mission Mountains, grizzlies exhibit seasonal differences in habitat use (Servheen 1983). The following habitats are used more than expected based on availability: in spring, low-elevation riparian zones and wet seeps; in summer, wet seeps and alpine slabrock; and in fall, riparian zones, wet seeps, wet meadows, and alpine slabrock. In spring, only agricultural lands were used less than expected; these areas were not completely avoided, but traversed at night. Timber, timbered shrubfield, and agricultural lands were used less than expected in summer. However, agricultural lands were important foraging sites in the fall, although daylight use was not recorded.

In the North and South Forks of the Flathead River drainage, four bears studied in 1979 were found to prefer snowchutes, ridgetops, and creek bottoms in spring, and shrubfields, slabrock, ridgetops, and creek bottoms in summer and fall (Zager et al. 1983). Timbered stringers between harvest units were also important because they were used as travel corridors and occasional daytime bedding areas. Harvest units and habitat affected by proximity of open, traveled roads were avoided throughout the active season, as was timber. Harvest units used by grizzlies were usually isolated from human disturbance (71% of units were along open secondary or closed roads), and provided cover (well-developed shrub layer, leave trees, or unit boundaries) within 50 m. Thus, use of harvest units was not based solely on food availability, but also on proximity of open roads and availability of escape cover. Zager et al. (1983) also suggested that fire suppression has had a negative impact on grizzly habitat and food production on mesic sites. Even-aged stands of second-growth forest with sparse understories offer poor habitat, and logging activities promoting production of berries and herbaceous plants may improve habitat conditions (see Peek et al. 1987).

However, the concept of improving habitat through silvicultural manipulation may only apply in areas where grizzly bear security is maximized (Mace and Manley 1993). Preliminary results in the South Fork of the Flathead suggest that once an area has been roaded and some stands treated, it will receive less use by adult females. Once total road densities exceed 2.0 mi/mi$^2$, use by all bears is predicted to decline.

Similarly, in the Cabinet Mountains, grizzlies used habitat 0-914 m from open roads less than expected in the spring and fall (Kasworm and Manley 1990). Habitat within 122 m of trails also received less use than expected, but this appeared to be a function of habitats available within that zone. Further, mean distances of grizzly locations from a seasonally closed road increased when the road was opened (from 655 to 1122 m).

Most bears used areas < 100 m from roads less than expected in the North Fork of the Flathead; many of these areas contained important bear foods (McLellan and Shackleton 1988). Roads have a detrimental effect on bears by improving accessibility, often rendering populations more vulnerable. In the North Fork, all 29 known or suspected deaths between 1979-1988 were due to legal or illegal hunting; most bears were shot from roads. Thus, roads may pose the greatest threat to grizzly habitat today (USDI:FWS 1993).

## MODELING RULES:

- Select all cover types except water, urban, and agricultural areas.
- From the above set, select polygons with mean road densities $\leq 2$ mi/mi$^2$; include all roads in the density calculation regardless of closure status.

## ASSUMPTIONS:

- Because diversity is the key to prime habitat, only human-influenced cover types (and water) should be excluded from the analysis. Certainly, a more complex model would assign higher values to individual habitats on a seasonal basis.
- Security from human disturbance is the limiting factor in the diverse Seeley-Swan, and road densities are an adequate measure of security. Other indicators of vulnerability, such as towns and campgrounds, will be accounted for by the road criterion.
- Although trails also increase accessibility, they can be excluded from road density calculations. Mace and Manley (1993) included trails in their calculation of unroaded areas within the cumulative adult female home range for the South Fork.
- Important habitats violating the road criterion should not be included in maps of predicted habitat; even if they are receiving use, they are not secure areas.

- Seedling/sapling types most likely to be used by grizzly bears are those farther from roads; thus, this cover type can be included in the modeling process.

- Denning habitat does not need separate consideration; because most dens are at higher elevations (2050-2500 m, Servheen and Klaver 1983), denning areas are likely to have low road densities, and thus will be included in maps of habitat.

- Lacking comprehensive data on roads in the 1930s, it is assumed that no areas had road densities >3 mi/mi$^2$ at that time.

# MOUNTAIN GOAT
*(Oreamnos americanus)*

## HABITAT:

Mountain goats inhabit the upper elevations of the Northern Rocky Mountains. Summer habitat is typically found at higher elevation than winter habitat (Rideout 1974, Smith 1976, Singer and Doherty 1985, Hayden 1989). Cliffs are especially important habitat components.

In the Sapphire Mountains, Rideout (1974) found that cliffs were used yearround; approximately half of goat locations were on steep slopes or cliffs. North and east aspects were used the most in summer and fall, while in winter, the relatively snow-free south and west aspects were used. Forested areas received the greatest use in July, August, and October, and meadows were used the most in summer and fall.

Similar trends were found in the Cabinet Mountains (Burleigh 1978), where rock, Douglas-fir/shrub, and subalpine habitat types were used throughout the year. Goats occupied subalpine grasslands as well as glacially carved basins and escarpments in summer. Cliffs were a component of all winter ranges, where south aspects predominated; rockland/talus types were used most in winter, followed by cirque basins.

Mountain goats preferred alpine forb meadows, forb-dominated outcrops, forested crops, subalpine fir/beargrass krummholz, and mineral lick habitat types in Glacier National Park (Singer and Doherty 1985). Although coniferous forest occupied 55% of the study area in the southern part of the park, less than 1% of goat locations were found within that type.

Winter ranges in the Bitterroots were characterized by steep broken terrain dominated by tiered cliffs; steep slopes, southerly exposures, and wind action contribute to excellent snow-shedding properties (Smith 1976). Between January and May, 94% of observations were between 4200-6500', while nearly all summer locations were >7300'. Cirques were used heavily in summer, fulfilling all habitat needs in that period by providing abundant lush forage, water, bedding, and escape terrain. Northerly and easterly aspects were used 67% of the time in summer; lush sedge/forb mats were available on these exposures. Subalpine fir or alpine larch overstory was present for 60% of summer locations.

In the Swan Range, cliff exposures of argillaceous rock outcrop with local Douglas-fir communities were prominently distributed on south and west facing slopes within spruce-fir forests (Chadwick 1973). Most goat habitat was found within the subalpine zone, 5000-9000'. Summer and winter ranges coincided in the Bunker Creek study area, although summer ranges occupied more area. Cliff and ledge types were typically lightly timbered in portions; they were used throughout the year, particularly during the early spring. In the summer, goats occupied cliffs, dry meadows, and ravine-wet meadows. Goats wintered on windblown ridgetops (roughly 7200') and cliffs. Use of various habitats seemed largely based on seasonal differences in forage palatability and accessibility. During the study, goats were displaced by road-building and blasting activities in the Bunker Creek vicinity.

MODELING RULES:

- Select barren, rocky woodland, grass, and shrub cover types.
- From the above set, select areas above 1845 m (6000').
- In addition, select all whitebark pine/Engelmann spruce/subalpine fir cover types, regardless of elevation.

ASSUMPTIONS:

- Summer and winter ranges were not treated individually. Winter ranges are likely to be a limiting factor in mountain goat distributions, and might be more appropriately handled separately. However, the model as described should adequately capture suitable winter ranges.
- The elevational cutoff of 6000' is an arbitrary definition selected to represent the range of elevations used in various studies. Because of the distribution of the habitat types selected above, modeling results are not overly sensitive to the elevational cutoff applied. Results match quite well with mountain goat habitat as delineated by FWP biologists (digital layer acquired May 1992, Bissell, pers. comm.).

## SHIRAS MOOSE
*(Alces alces shirasi)*

HABITAT:

Shiras moose made local range extensions in Montana from 1940-1970 (Stevens 1971, in Peek 1974), and have since continued to expand their range (Bissell, pers. comm.). Key elements of Shiras moose habitat in northwestern Montana (as described for the Yaak region) include forage, hiding cover, overhead cover, and aquatic sites (Costain 1989). Seasonal differences in habitat use are apparent. For example, moose select for lower elevations in winter, then typically disperse to higher elevations in summer (Pierce and Peek 1984, Matchett 1985, Costain 1989, Langley 1993). Year-round, moose in the Yaak selected for flat and rolling terrain, avoiding slopes >45% (Matchett 1985). In the North Fork of the Flathead, also in northwestern Montana, Langley (1993) observed selection for gentler slopes as well.

Selection for forest types also differs by season. Pierce and Peek (1984) found old-growth grand fir/Pacific yew stands to be critical winter habitat in north-central Idaho, while in summer, use was heaviest in pole stands and open areas (clearcuts and lakes). Use of old growth was greater than expected; about 50% of fall, winter, and spring locations were in old growth. In the Yaak, Matchett (1985) found that logged areas were used more in early winter, while dense timber (often in draws and stream bottoms) received more use in mid-late winter. Year-round, moose selected for clearcuts, logged areas less than 12 ha, and areas logged 15-30 years ago. Although use of unlogged sites was less than expected based on availability, moose were found in these areas more than 50% of the time. Costain (1989) reported that moose in the Yaak preferred sapling stands in fall, mild winters, and spring, and mature timber and larger saplings in severe winters. Pole stands were avoided in all seasons but summer, and no selection was found for seedlings, large mature, or old-growth stands. Moose strongly selected for habitats with abundant forage except in hot summer conditions or periods of deep snow, when they retreated to forest stands providing a thermal umbrella. In the North Fork of the Flathead, Singer (1979) found a preference for old-growth spruce in winter; this type offered reduced snow depths, excellent overhead cover, plentiful forage, and snow-free travel along the river. Langley (1993) studied 29 cows in the North Fork, 21 of which migrated to higher elevations in summer and 8 which used the same area year-round. She found that the home ranges of nonmigratory moose and the summer ranges of migratory moose contained more marsh and sapling cover types than expected, while the winter ranges of migratory animals had more conifer cover and greater lengths of permanent river.

Streamside complexes of willow bottoms and conifers are an important component of winter range (Smith 1962, Peek 1974), and proximity to water is notable year-round. In the Yaak, Matchett (1985) found selection for areas less than 100 m from water in all seasons, and Costain (1989) found moose in habitats adjacent to streams, ponds, or swampy areas 41-52% of the time. Costain (1989) reported strong selection against sites >1500' from water in winter and spring, and selection for sites <300' in harsh winters and summer, as well as a significant increase in use of stream bottoms and draws May-October. Stream bottoms, draws, and swamps within stands of mature timber with good canopy closure provided hiding and thermal cover, and were heavily used for summer feeding. Such areas, along with aquatic feeding sites and calving sites, were identified as key habitat components in the Yaak (Costain 1989). Good calving sites provide dense hiding cover and proximity to water and forage; they are usually in mature/old-growth stands >150 ac (Costain 1989). Matchett (1985) found that cows with calves used older, wetter, more thickly vegetated sites than those without. Langley (1993) noted several variables suggesting selection for heavy cover at calving sites.

In sum, Matchett (1985) suggested that productive habitat for Shiras moose is best provided by a mosaic of small, 15-30 year-old cuts interspersed with mature, closed canopy timber. This juxtaposition of sapling stands providing forage and older forest offering hiding and thermal cover is best examined at a landscape scale; thus, Shiras moose is a natural choice for habitat modeling using a GIS.

MODELING RULES:

- Create a 150 m buffer around all lakes, marshes, and intermittent and perennial streams.

- Within the buffer, select for seedling/sapling and mature/overmature forest types, along with all shrub and broadleaf types.

- Narrow the selected set further by eliminating stands with mean slope >50%.

- For individual polygons, calculate the percentage of area predicted to be habitat.

ASSUMPTIONS:

- The above modeling rules will adequately represent winter and summer habitat for Shiras moose.

- Excluding areas >150 m from water will not lead to a gross underestimation of moose habitat in the Seeley-Swan. Areas in closest proximity are assumed to be of highest quality. In addition, because hydrographic features in the study area are narrowly spaced, a 150 m buffer around all features includes a high proportion of the study area.

- Including seedling stands will not lead to a gross overestimation of habitat. Although no study identified selection for seedling types, I am not able to separate seedling and sapling stands due to the mapping rules used in the 1930s.

- Selection was found to be strongest for areas within 100 m of water; I conservatively opted to extend the buffer to 150 m.

## APPENDIX D:
## Sample habitat program in Arc Macro Language (AML, ESRI 1991), ARC/INFO Version 6.1.1

```
/* AML to predict peregrine falcon habitat in the 1930s and 1990s
/* 2 June 1994, Melissa Hart

w /scratch/mhart/d.wild
&echo &on
&watch falper.wat
w d.falper

/* Select out areas with slope greater than or equal to 150%, regiongroup them,
/* select only the regions at or below 2615 m and 5 acres or larger (22 pixels).
/* Then run zonalstats on the selected areas using the DEM to find the change
/* elevation for each region. Change in elevation is used to approximate cliff
/* height, and should be at least 90 m. In this case, all regions satisfy that
/* criterion, so no need to narrow the selected set.

/* The same steep areas are identified as nesting habitat for all time periods
/* and minimum mapping units, for fairly obvious reasons.

grid
steep = con(/wren/mhart/d.topo/slopeint ge 150, 1, 0)
steepreg = regiongroup(steep)
steepelev = con(/wren/mhart/d.topo/swandem_alb le 2615, steepreg, 0)
steeparea = select(steepelev, 'value gt 1 and count ge 22')
steeptable = zonalstats(steeparea, /wren/mhart/d.topo/swandem_alb, all)
q
additem steeparea.vat steeparea.vat elevchange 4 10 f 3
&data arc info
arc
SEL STEEPAREA.VAT
RELATE STEEPTABLE BY VALUE
CAL ELEVCHANGE = $1RANGE
Q STOP
&end
```

```
/* Next, calculate euclidean distances from lakes and streams, and overlay
/* selected steep areas with those distances using zonalstats to identify
/* minimum distances from lakes and streams for each steep area. Steep
/* areas should not be more than a mile (1610 m) from a water body; here
/* again, all areas meet the criterion.

reselect /wren/mhart/d.wild/d.gavimm/lakeclip lakeonly poly
reselect code = 300
~
n
n
polygrid lakeonly lakegrid lakeonly-id
30
y
grid
lake_dist = int(eucdistance(lakegrid) + 0.5)
stream_dist = int(eucdistance(../d.asctru/streamgrid) + 0.5)
lake_disttab = zonalstats(steeparea, lake_dist, all)
stream_disttab = zonalstats(steeparea, stream_dist, all)
q
additem steeparea.vat steeparea.vat lakedistmin 4 10 f 3
additem steeparea.vat steeparea.vat streamdistmin 4 10 f 3
&data arc info
arc
SEL STEEPAREA.VAT
RELATE LAKE_DISTTAB BY VALUE
CAL LAKEDISTMIN = $1MIN
RELATE STREAM_DISTTAB BY VALUE
CAL STREAMDISTMIN = $1MIN
RES LAKEDISTMIN LE 1610 OR STREAMDISTMIN LE 1610
ASE
Q STOP
&end

/* Calculate euclidean distances from steep areas; these will be used to select
/* foraging areas in the upcoming AMLs. Create a grid with a value of 1 for all
/* cliff pixels, also to be used in upcoming AMLs as a means of calculating the
/* percentage of a polygon occupied by cliffs.

grid
steep_dist = int(eucdistance(steeparea) + 0.5)
steepval = con(steeparea gt 0, 1)
q
```

```
/* Add items to master databases, copy to directory d.falper, run individual AMLs
/* (see following pages) within this master program, copy results back to master
/* databases, and delete intermediate files.

w ..
additem wildtable30 wildtable30 falper 2 4 b
additem wildtable30 wildtable30 falperarea 4 10 f 3
additem wildtable90 wildtable90 falper 2 4 b
additem wildtable90 wildtable90 falperarea 4 10 f 3
additem wildtable16 wildtable16 falper 2 4 b
additem wildtable16 wildtable16 falperarea 4 10 f 3
copyinfo wildtable30 d.falper/wildtable30_falper
copyinfo wildtable90 d.falper/wildtable90_falper
copyinfo wildtable16 d.falper/wildtable16_falper
w d.falper
&r falper30
&r falper90
&r falper16
w ..
copyinfo d.falper/wildtable30_falper wildtable30_falper
copyinfo d.falper/wildtable90_falper wildtable90_falper
copyinfo d.falper/wildtable16_falper wildtable16_falper
&data arc info
arc
SELECT WILDTABLE30
RELATE WILDTABLE30_FALPER BY VALUE
CAL FALPER = $1FALPER
CAL FALPERAREA = $1FALPERAREA
RELATE
SELECT WILDTABLE90
RELATE WILDTABLE90_FALPER BY VALUE
CAL FALPER = $1FALPER
CAL FALPERAREA = $1FALPERAREA
RELATE
SELECT WILDTABLE16
RELATE WILDTABLE16_FALPER BY VALUE
CAL FALPER = $1FALPER
CAL FALPERAREA = $1FALPERAREA
RELATE
SELECT WILDTABLE30_FALPER
ERASE WILDTABLE30_FALPER
Y
SELECT WILDTABLE90_FALPER
```

```
ERASE WILDTABLE90_FALPER
Y
SELECT WILDTABLE16_FALPER
ERASE WILDTABLE16_FALPER
Y
Q STOP
&end
&watch &off


/*  first subroutine
```

---

FALPER30.AML: creates map of predicted habitat for the peregrine falcon in the 1930s based on a vegetation layer with 16 ha minimum mapping unit.

---

```
copy /wren/mhart/d.presettle/swan30grida
additem swan30grida.vat swan30grida.vat tempwild 2 4 b

/*  Select cover types used for foraging:  water, grass, and agriculture.  For the
/*  selected foraging areas, find euclidean distances from cliffs using zonalstats.
/*  Map foraging areas by selecting polygons with a minimum distance less than 10
/*  miles (16,100 m).  Merge the foraging and nesting areas into one grid.  Then
/*  identify the polygons in the master vegetation file, swan30grida, containing
/*  the cliff areas -- again by using zonalstats -- and note the sum of the cliff
/*  pixels within each vegetation polygon.

&data arc info
arc
SEL SWAN30GRIDA.VAT
RES STD_COV = 1 XOR STD_COV = 4 XOR STD_COV = 8
CAL TEMPWILD = 1
ASE
Q STOP
&end
grid
forage30 = con(swan30grida.tempwild == 1, swan30grida, 0)
buildvat forage30
steep_disttab30 = zonalstats(forage30, steep_dist)
q
additem forage30.vat forage30.vat mincliffdist 4 10 f 3
&data arc info
arc
```

```
SEL FORAGE30.VAT
RELATE STEEP_DISTTAB30 BY VALUE
RES $1VALUE = VALUE
CAL MINCLIFFDIST = $1MIN
ASE
Q STOP
&end
grid
forage30dist = con(forage30.mincliffdist le 16100 and forage30.value gt 0, 2)
nestforage30 = merge(steepval, forage30dist)
steepvaltab30 = zonalstats(swan30grida, steepval, sum)
q

/* Write results to the intermediate file so that they can be transferred to the
/* master database: identify foraging polygons in the database with a value of 1
/* and an area of 1, and nesting polygons with a value of 1 and a percentage
/* value (>0) for area. (The percentage is equal to the number of cliff pixels
/* within each polygon identified to contain some proportion of nesting habitat.
/* Note that the values are the same for both nesting and foraging habitat; I'm
/* assuming that it will be sufficient to show the difference between the two on
/* the map (which is possible using the file nestforage30), and that a value of 1
/* must be entered for all species -- it should indicate predicted presence or
/* absence, not type of habitat, because it will be used to sum species richness
/* for each polygon.

&data arc info
arc
SEL WILDTABLE30_FALPER
RELATE FORAGE30.VAT BY VALUE
RES $1VALUE = VALUE
RES VALUE GT 0 AND $1MINCLIFFDIST LE 16100
CAL FALPER = 1
CAL FALPERAREA = 1
ASE
RELATE
RELATE STEEPVALTAB30 BY VALUE
RES $1VALUE = VALUE
CAL FALPER = 1
CAL FALPERAREA = $1SUM / COUNT
ASE
Q STOP
&end
```

```
/* second subroutine, essentially identical to first except for vegetation file
/* manipulated
```

---

FALPER90.AML: creates map of predicted habitat for the peregrine falcon in the 1990s based on a vegetation layer with 2 ha minimum mapping unit.

---

```
copy /wren/mhart/d.veg/swanmerge
additem swanmerge.vat swanmerge.vat tempwild 2 4 b

&data arc info
arc
SEL SWANMERGE.VAT
RES STD_COV2 = 1 XOR STD_COV2 = 4 XOR STD_COV2 = 8
CAL TEMPWILD = 1
ASE
Q STOP
&end
grid
forage90 = con(swanmerge.tempwild == 1, swanmerge, 0)
buildvat forage90
steep_disttab90 = zonalstats(forage90, steep_dist)
q
additem forage90.vat forage90.vat mincliffdist 4 10 f 3
&data arc info
arc
SEL FORAGE90.VAT
RELATE STEEP_DISTTAB90 BY VALUE
RES $1VALUE = VALUE
CAL MINCLIFFDIST = $1MIN
ASE
Q STOP
&end
grid
forage90dist = con(forage90.mincliffdist le 16100 and forage90.value gt 0, 2)
nestforage90 = merge(steepval, forage90dist)
steepvaltab90 = zonalstats(swanmerge, steepval, sum)
q
```

```
&data arc info
arc
SEL WILDTABLE90_FALPER
RELATE FORAGE90.VAT BY VALUE
RES $1VALUE = VALUE
RES VALUE GT 0 AND $1MINCLIFFDIST LE 16100
CAL FALPER = 1
CAL FALPERAREA = 1
ASE
RELATE
RELATE STEEPVALTAB90 BY VALUE
RES $1VALUE = VALUE
CAL FALPER = 1
CAL FALPERAREA = $1SUM / COUNT
ASE
Q STOP
&end


/* third subroutine, also essentially identical to first and second
```

---

FALPER16.AML: creates map of predicted habitat for the peregrine falcon in the 1990s based on a vegetation layer with 16 ha minimum mapping unit.

---

```
copy /wren/mhart/d.veg/d.std16ha/std16hareg
additem std16hareg.vat std16hareg.vat tempwild 2 4 b


&data arc info
arc
SEL STD16HAREG.VAT
RES STD_COV = 1 XOR STD_COV = 4 XOR STD_COV = 8
CAL TEMPWILD = 1
ASE
Q STOP
&end
grid
forage16 = con(std16hareg.tempwild == 1, std16hareg, 0)
buildvat forage16
steep_disttab16 = zonalstats(forage16, steep_dist)
q
additem forage16.vat forage16.vat mincliffdist 4 10 f 3
```

```
&data arc info
arc
SEL FORAGE16.VAT
RELATE STEEP_DISTTAB16 BY VALUE
RES $1VALUE = VALUE
CAL MINCLIFFDIST = $1MIN
ASE
Q STOP
&end
grid
forage16dist = con(forage16.mincliffdist le 16100 and forage16.value gt 0, 2)
nestforage16 = merge(steepval, forage16dist)
steepvaltab16 = zonalstats(std16hareg, steepval, sum)
q


&data arc info
arc
SEL WILDTABLE16_FALPER
RELATE FORAGE16.VAT BY VALUE
RES $1VALUE = VALUE
RES VALUE GT 0 AND $1MINCLIFFDIST LE 16100
CAL FALPER = 1
CAL FALPERAREA = 1
ASE
RELATE
RELATE STEEPVALTAB16 BY VALUE
RES $1VALUE = VALUE
CAL FALPER = 1
CAL FALPERAREA = $1SUM / COUNT
ASE
Q STOP
&end

/* finally...
```